THÈSE

En vue de l’obtention du

DOCTORAT DE L’UNIVERSITÉ DE TOULOUSE

Délivré par l'Université Toulouse 3 - Paul Sabatier

Présentée et soutenue par

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Le 28 janvier 2019

Impacts environnementaux de l'exploitation pétrolière en Amazonie équatorienne: de l'étude spatiale de la vulnérabilité à l'évaluation du risque

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Ecole doctorale : **SDU2E - Sciences de l'Univers, de l'Environnement et de l'Espace**

Spécialité : **Ecologie fonctionnelle**

Unité de recherche :
ECOLAB - Laboratoire d'Ecologie Fonctionnelle et Environnement

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i. Acknowledgements

Albert Einstein’s quotes about time relativity come to mind and make sense, probably in another manner but they do. Sun goes up and down, again and again, time marches on, while your research has yet not advanced... It seems that during a long lasting trip, such as the Ph.D. research time, motivation and perseverance are probably the main qualities a Ph.D. student must develop. I want to thank my supervisors Dr. Arnaud Elger and Dr. Mehdi Saqalli for their motivation and the scientific and academic knowledge they have passed on to me. Their support was essential for the completion of this work. I also want to thank Sylvia Becerra and Laurence Maurice for providing me the opportunity to participate in MONOIL project. It was a good and interesting learning experience that I will remember for the rest of my life. I also thank for the personal, as well as, the scientific exchanges with MONOIL PhD students, Guilhem, Fausto and Fiorella, post-doc Nicolas and co-supervised interns Barbara, Edna, Marine, Pierre, for without their advices and support this work would have taken longer to progress.

I would also like to thank the team at Ecolab, the motivational support and advices of Franck and Eric, the assistance with GIS and computer systems by Hughes and Julien, and Annick always there to give a hand for any problem presented. In addition, to my colleagues for their invaluable moral support and some suggestions during the thesis, Deoni and Steve, Amine, Sylvain, Theo, Adrien, Vivian, and the list could extend vastly. More than anything the daily canteen lunch time that we seldom skipped. I also thank to the people in Ecuador who support the field trips, the Ecuadorian official from PRAS and the Amazon Defense Front, this work is dedicated to all those who try to improve the living quality and life standards of the Amazonian people.

At last but not least I want to thank my parents for believing and supporting every week for four years my discussions about this topic. Sometimes just listening to what I am doing without even knowing what I was talking about. I thank also the patience and caring Dana have had, I sincerely do not want to be in her shoes but I appreciate the love withstanding some long nights of hard work. Thanks to Yadira the “spider” and Julio the “escargot” that were always gracefully providing their company as well. This last line probably nobody will understand, but anyways, I hope this journey is not finishing but just beginning…

Good read to you!

ii. Résumé

L'Équateur est le 5ème producteur de pétrole d'Amérique latine. La plupart des réserves se trouvent sous le nord-est de l'Amazonie équatorienne (NEA), représentant 15% de l'ensemble du pays, mais englobant une grande diversité biologique et culturelle. La production de pétrole et de gaz génère des déchets toxiques susceptibles de polluer l'environnement. La méthodologie a été définie pour évaluer les aléas et la vulnérabilité environnementale en tant que composantes indépendantes du risque, en utilisant des méthodes indicielles et des outils de hiérarchisation. Ensuite, ils ont été combinés à l'aide de méthodes de superposition spatiale. La qualité des données publiques utilisées dans cette étude a constitué une difficulté. Dans ce contexte, le premier objectif était de déterminer les volumes d'hydrocarbures déversés accidentellement dans des blocs pétroliers bien documentés. Ensuite, des volumes de déversements estimés ont été attribués aux blocs mal documentés pour obtenir une carte homogène. Le deuxième objectif consistait à cartographier les principales émissions atmosphériques associées aux torchères, c'est-à-dire les gaz à effet de serre (CO_2, CH_4) et les particules de noir de carbone (BC). Le troisième objectif était d'évaluer la vulnérabilité potentielle du patrimoine naturel à l'échelle régionale à l'aide de proxys tels que le statut de protection et l'occupation des sols. Le quatrième objectif consistait à illustrer l'approche proposée pour l'évaluation des risques en évaluant le potentiel de contamination des eaux souterraines à partir des fosses de stockage de résidus d'hydrocarbures. Les principaux résultats indiquent 10 000,2 t (909,1 t.an^{-1}; SD = 1219,5) de pétrole déversé accidentellement dans la NEA restrictif à la période 2001-2011 (inclus, soit 11 ans), selon les événements enregistrés. Cependant, une augmentation de 54.8% a été constatée lors de l'extrapolation des taux de déversement des blocs pétroliers bien documentés aux blocs mal documentés. La précision des prévisions spatialisées a été de 32 à 97%. Les gaz brûlés au cours de la période 2003-2012 se sont élevés à 7,6 Gm^3 (760 Mm^3.yr^{-1}), ce qui correspond à des valeurs allant de 3,7 à 4,5 kt.an^{-1} BC. Les hydrocarbures dans les fosses de stockage ont été estimés à 49 436,4 t. Plusieurs cartes résultent de cette thèse. Les émissions spatialisées indiquent que les déversements et les émissions des torchères sont plus fréquents dans les agglomérations de Joya de los Sachas, Dayuma et Shushufindi. Les cartes de vulnérabilité du patrimoine naturel indiquent que 42% de la surface du territoire est hautement vulnérable, à l'est de la zone d'étude. La vulnérabilité des eaux souterraines est faible à moyenne dans la plupart des zones. En outre, l'exemple envisagé pour l'évaluation des risques liés aux eaux souterraines et aux fosses non étanchéifiées indique que les impacts potentiels les plus importants sont localisés au niveau des agglomérations de Nueva Loja, Tarapoa et Shushufindi. La qualité des données publiques disponibles a été jugée acceptable. Par exemple, en comparant nos estimations des émissions atmosphériques avec d'autres estimations indépendantes, une différence de 2,5 fois au maximum a été trouvée. La précision de la répartition spatiale des déversements accidentels a révélé une méthodologie prometteuse pour améliorer la cartographie des aléas. L'évaluation de la vulnérabilité a montré que les composantes du patrimoine naturel permettent de construire des indices de vulnérabilité à l'échelle régionale, l'occupation des sols étant significativement corrélée à la richesse spécifique et les aires protégées étant conservées efficacement sur le long terme, véhiculant ainsi une information sur l'intégrité écologique. De plus, il n'y a que 8,8% d'incongruence spatiale entre les deux indices. La cartographie de la vulnérabilité des eaux souterraines a révélé des lacunes dans les connaissances discutées. Certains seuils de distance ont été proposés pour sélectionner les sites de validation dans les études futures. En conclusion, les estimations et les cartes obtenues peuvent s'avérer utiles pour la surveillance de la sécurité et la sûreté des installations, la responsabilisation des institutions publiques et l'aménagement du territoire afin de réduire les risques futurs.

Mots-clés : inventaire des émissions, activités pétrolières, Amazonie, analyse spatialisée, hotspots de pollution, analyse du risque, divulgation et qualité des données.

ii. Abstract

Ecuador is the 5th oil producer in Latin America. Most of crude oil reserves lie beneath the north-eastern Ecuadorian Amazon (NEA), representing 15% of the entire country, yet encompassing high biodiversity and cultural heritage. Crude oil and gas production generate toxic wastes potentially polluting the environment. The methodology was set to evaluate hazards and environmental vulnerability, using score indexes and rankings, as independent components of risk. Then, they were combined using spatial overlay methods. An observed hindrance for risk analysis was the quality of public data that were used in this study. In this context, the first aim was to determine accidental oil spill volumes in well-documented oil blocks. Then, putative spill volumes were allocated to poorly-documented oil blocks to obtain a homogeneous map. The second aim was to map key atmospheric emissions associated to gas flaring, i.e., greenhouse gas (CO_2, CH_4) and black carbon (BC) particles. The third aim was to assess the potential vulnerability of natural heritage using regional scale proxies such as protection status and land use. Finally, the fourth aim was to exemplify the presented risk assessment approach by evaluating total petroleum hydrocarbons (TPH) potentially flowing to groundwater from oil pits. Main results indicate 10,000.2 t (909.1 $t.yr^{-1}$; SD = 1,219.5) oil spilled in the NEA during the 2001-2011 period (11 years), according to recorded events. However, a 54.8% increase was found when extrapolating spill rates from well-documented oil blocks to poorly-documented ones. Spatial prediction accuracy ranged from 32 to 97%. Gas flared amounted to 7.6 Gm^3 (760 $Mm^3.yr^{-1}$), equivalent to a range of 3.7 – 4.5 $kt.yr^{-1}$ BC, during 2003-2012 lapse. Total petroleum hydrocarbons in unlined oil pits was estimated to 49,436.4 t. Several maps resulted from this thesis. Spatial emissions indicate spills and gas flaring are occurring at higher rates in settlements of Joya de los Sachas, Dayuma and Shushufindi. The natural heritage vulnerability maps indicated 42% of highly vulnerable surface at the most eastern side of the studied area. Groundwater vulnerability was low to medium in most areas; furthermore the example considered for risk assessment of groundwater and unlined oil pits, indicated highest potential impacts in settlements of Nueva Loja, Tarapoa and Shushufindi. Publicly available data quality was found to be acceptable. For instance, when comparing airborne emission estimates with some other independent estimates only 2.5-fold difference was found at most. Spatial allocation accuracy of oil spills showed promising methodology for improving hazard mapping. Vulnerability assessment indicated natural heritage proxies to be suitable for building vulnerability indexes at regional scale as land use is significantly correlated to species richness, and protected areas are efficiently conserved in the long term, thus conveying some information on ecological integrity. Moreover, there was only 8.8% of spatial incongruence between the two proxies. Groundwater vulnerability mapping indicated gaps in knowledge that were discussed; some distance thresholds were proposed to select validation sites in future studies. In conclusion, estimates and maps obtained may be valuable for safety and security monitoring, accountability of public institutions and land use planning to lessen future risks.

Keywords: emissions inventories, oil activities, the Amazon, spatial analysis, pollution hotspot, risk assessment, public disclosure and data quality.

iii. Abbreviations

ADF	Amazon Defense Front
APG	Associated petroleum gas
API	American Petroleum Index
ARCH	Agency of Hydrocarbons Regulation Agency
BC	Black carbon
BDv	Biodiversity value
BVI	Biodiversity Vulnerability Index
CDIAC	Carbon Dioxyde Information Analysis Center
CE	Gas conservation efficiency factor
CEPE	National Petroleum Corporation
CH_4	Methane
CO_2	Carbon dioxide
DEM	Digital Elevation Model
DMSP-OLS	Night-time imagery operational line scan system
DNH	National Hydrocarbons Directive
DRASTIC	Groundwater vulnerability model
ECD	Energy Census Data
EF	Emission factor
EIV	Ecological Integrity Index
EPA	Environmental Protection Agency of the United States
ERA	Environmental Risk Assessment
GAD	Autonomous decentralized government
GFFR	Global Gas Flaring Reduction Partnership
GHG	Greenhouse gases
Gi	Getis-Ord index
GIS	Geographic information systems
GWP	Global warming potential
IARC	International Agency for Research on Cancer
IEE	Ecuadorian Spatial Institute
IERAC	Ecuadorian Agrarian Institute for Agrarian Reform and Colonization
IGM	Geographical Military Institute
INEC	National Institute of Statistics and Census of Ecuador
INHAMI	Institute of Meteorology and Hydrology of Ecuador
IOPCF	International Oil Pollution Compensation Fund
IPCC	Intergovernmental Panel for Climate Change
ITCZ	Intertropical Convergence Zone

IUCN	International Union for Conservation of Nature
LOTAIP	Ecuadorian Transparency and Access to Public Information Law
LU	Land use
MAE	Ministry of the Environment of Ecuador
MAGAP	Ministry of Agriculture, Livestock, Aquaculture and Fisheries
MDR	Maximum Distance Range
MEM	Ministry of Energy and Mines
MONOIL	Research Program for Monitoring Oil Activities in Ecuador
MSS	Ministry of Strategic Sectors
NEA	Northern-Ecuadorian Amazon
NOAA	National Oceanographic and Atmospheric Association
OCP	Heavy Crudes Pipeline
PA	Protected areas
PAH	Polycyclic Aromatic Hydrocarbons
PEB	Potential loss of ecological integrity and biodiversity index
PM	Particulate matter
PRAS	Program of Social and Environmental Remediation
PS	Protection status
PSB	Socio-Bosque Program
RA	Risk Assessment
RAHOE	Regulation of oil activities in Ecuador
SNAP	Protected Areas Natural Heritage
SOTE	Trans Ecuadorian Pipeline System
SUIA	System of Environmental Information
TPH	Total petroleum hydrocarbons
UN	United Nations
UNDATA	United Nations databases
UNESCO	United Nations Educational, Scientific and Cultural Organization
OP	Unlined oil pits
VIIRS	Visible infrared imaging radiometer suite
ZITT	Intangible zones for the Tagaeri-Taromenane tribes

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“Maps are like campfires – everyone gathers around them, because they allow people to understand complex issues at a glance, and find agreement about how to help the land.”

— JP Benzécri

1. **Introduction générale (version abrégée en français)**

1.1. Risque associé aux activités pétrolières

Le domaine du risque a des racines aussi anciennes que le comportement humain lui-même. Les premières réflexions sur le risque sont liées au fait que l'incertitude des événements futurs a été laissée à la volonté de Dieu (Rausand, 2011). Il existe des risques non mesurables et mesurables (Adger, 2006). Le premier peut être envisagé avec une analyse qualitative tandis que le second peut être traité avec des métriques permettant une quantification de ces risques (SRA, 2015). Les deux ont pour objet de fournir des informations pour la prise de décisions appropriées sur un plan d'action futur. Les risques peuvent être de plusieurs types : naturels, écologiques, sociaux, politiques, technologiques, financiers… (Gleyze, 2002). Ces risques ne peuvent pas être complètement évités, mais les humains disposent de moyens techniques améliorés pour les gérer en réduisant leur occurrence et en atténuant leurs conséquences lorsqu'ils se présentent.

1.1.1. Évaluation des risques (ER)

Le risque peut être mesuré en fonction de l'aléa et de la vulnérabilité comme montre l'Equation 1 (Jörn Birkmann and Wisner, 2006; Gleyze, 2002; Kazakis and Voudouris, 2015; Metzger et al., 2008). Un aléa est « le dommage potentiel d'un événement physique, d'un phénomène ou d'une activité humaine pouvant causer des pertes de vies humaines ou des blessures, des dommages matériels, des perturbations sociales et économiques ou une dégradation de l'environnement » (Nations Unies, 2004). Il peut être mesuré ou abordé de manière quantifiable par sa fréquence, son intensité ou sa probabilité d'occurrence (Gleyze, 2002).

La vulnérabilité est la manière dont un aléa peut nuire à un élément représentant un enjeu particulier, en fonction de ses propriétés intrinsèques et spécifiques (Berry et al., 2006). Cet élément qui pourrait être perdu est considéré comme important par la société qui tire parti de son existence (Metzger et al., 2008). Les enjeux environnementaux peuvent être constitués par la biodiversité, ainsi que par les écosystèmes et les services qu'ils fournissent (Barraza et al., 2018; Füssel, 2007; Metzger et al., 2006; Rahman, 2008). La vulnérabilité comprend trois composantes :

(1) La sensibilité, définie comme les caractéristiques intrinsèques du système qui le rendent sensible à la dégradation (Metzger et al. 2008).

(2) La résilience, qui est la capacité à rétablir la fonctionnalité du système après une perturbation ou un stress (Walker et al. 2015).

(3) L'exposition, qui correspond à la concomitance, spatiale et temporelle, avec l'aléa (GIEC, 2012; Nations Unies, 2004).

L'aléa et la vulnérabilité peuvent dans un premier temps être analysés comme des composantes séparées, qui pourront ensuite être utilisées pour définir le risque (Birkmann, 2011). Cette thèse propose une évaluation des risques pour les enjeux environnementaux (ERE) dans un contexte d'activités pétrolières. L'approche consiste en une méthode de couplage d'index, par une analyse de superposition quantitative et spatialisée, résumée dans l'équation 1:

$$Risque = \sum_{i=1}^{n} (Ap_i \times Aw_i) \times \sum_{j=1}^{m} (Vp_j \times Vw_j) \qquad \text{(Eq.1)}$$

<u>Où</u> :

n : nombres de paramètres constitutifs de l'aléa

Ap_i : valeur du paramètre i caractérisant l'aléa

Aw_i : poids donné au paramètre i dans la caractérisation de l'aléa

m : nombre de paramètres constitutifs de la vulnérabilité

Vp_j : valeur du paramètre j caractérisant la vulnérabilité

Vw_j : poids donné au paramètre j dans la caractérisation de la vulnérabilité

L'aléa comme la vulnérabilité peuvent en effet chacun être définis à partir de plusieurs paramètres constitutifs, pouvant correspondre à une quantité ou à une valeur qualitative susceptible d'être classée. Comme indiqué dans l'équation, chacun de ces paramètres peut être pondéré, de manière à quantifier l'importance du paramètre dans la valeur globale d'aléa ou de vulnérabilité. Si cette importance est inconnue, la pondération peut être ignorée ou considérée comme uniforme pour tous les paramètres (Cutter, Mitchell et Scott, 2000).

Les facteurs conditionnant la résilience du système peuvent être pris en compte dans cette équation comme paramètres constitutifs de la vulnérabilité. En revanche il n'est pas nécessaire de se référer explicitement à l'exposition. Celle-ci est en effet prise en compte de manière implicite : le risque résultant du produit de l'aléa et de la vulnérabilité, il s'annule dès lors que l’un d’entre eux est égal à zéro. En d’autres termes, dans le cadre d’une approche spatialisée, le risque est absent dans les zones où il n’y a pas de chevauchement entre l’aléa et la vulnérabilité.

1.1.2. Evaluation du risque environnemental pour les activités pétrolières

Au cours de la chaîne de production de pétrole brut, plusieurs produits résultant de ces activités peuvent (accidentellement ou intentionnellement) être rejetés dans l'environnement immédiat (Jernelöv, 2010). Ils peuvent constituer un danger pour les compartiments de l’hydrosphère, lithosphère et atmosphère, potentiellement constituent un danger pour les êtres vivants qu’y résident.

Les rejets liés aux activités pétrolières, tels que les déversements de pétrole, de saumure et de résidus de mine (constitués notamment d’hydrocarbures et de métaux lourds), ou encore le brûlage des gaz, libérant des particules, notamment de noir de carbone, et des gaz (CO_2, CH_4…), peuvent avoir des effets toxiques pour l’homme et pour les autres espèces et exacerber les changements climatiques. Les rejets intentionnels ou accidentels de contaminants liés au pétrole brut ont entraîné dans de nombreuses régions une pollution à long terme de l'air, des sols, des eaux de surface et des eaux souterraines (Barraza et al., 2017; Wernersson, 2004). Par exemple, les plantations de cacao équatoriennes présentent de fortes concentrations de cadmium (Barraza et al., 2017) et les hydrocarbures issus du pétrole ont montré des effets toxiques pour certains animaux aquatiques à Rio Rumiyacu, Dayuma, Équateur (Wernersson, 2004).

1.2. L'Amazonie équatorienne du nord-est en tant qu'étude de cas

1.2.1. Aperçu général et importance de la zone d'étude

L'Équateur est le cinquième producteur de pétrole en Amérique latine (OCDE, 2016). L’Équateur produit depuis cinq décennies du pétrole brut, principalement dans le nord-est de l’Amazonie équatorienne (NEA). La NEA est toutefois l'un des plus importants « hotspot » de biodiversité au monde (Bass et al. 2010) et abrite également huit populations humaines autochtones. Il s’agit historiquement d’un important cas d’étude environnemental car c’est un lieu qui a attiré l’attention de nombreux scientifiques depuis des décennies pour des recherches sur les systèmes socio-écologiques (Mena, 2008; Messina et Walsh, 2005; Walsh, et al. 2008) et sur la théorie de la complexité (Walsh et al. 2008). Cela tient notamment au contexte de forte déforestation à une échelle de temps récente (Messina, et al, 2006; Sierra, 2000; Messina et al.

2008). Ces pressions résultent du développement induit par le pétrole, qui a largement influencé l'ouverture des routes, qui emmène l'immigration et, par conséquent, le changement d'utilisation des sols (Baynard et al. 2013).

1.2.2. Mesures de conservation de la nature

Il est probablement impossible d'expliquer les mesures de conservation de la nature mise en œuvre dans la NEA sans expliquer le processus de colonisation impulsé par les réserves de pétrole découvertes sous la forêt amazonienne. Deux réformes agraires, en 1964 et 1973, ont encouragé la colonisation des terres forestières dans la NEA ; les colons engagés dans le défrichement des forêts s'assurant le titre de propriété des terres (Holland et al. 2014). En réaction à la dégradation de l'environnement, plusieurs sites du patrimoine naturel ont fait l'objet de mesures de protection par le Gouvernement Central, par les gouvernements locaux autonomes (GAD), et par des acteurs communautaires ou privés (MAE, 2015). Ces zones de conservation ont pour but de réglementer les activités socio-économiques, notamment via la protection des enjeux environnementaux sous la forme d'aires protégées (AP).

1.2.3. Activités pétrolières dans la NEA

Dans la NEA, la production de pétrole brut a commencé en 1960, apportant des revenus élevés à l'économie équatorienne. Malgré les nombreux avantages économiques des exportations de pétrole brut, ces activités ont entraîné des rejets de pollution accidentels ou intentionnels. Ces activités extractives sont susceptibles de nuire à l'environnement et aux communautés locales exposées, qui ont déclaré ne pas bénéficier de ces revenus (Buccina et al., 2013; San Sebastián and Hurtig, 2005; SENPLADES, 2013). Cela a abouti au tristement célèbre procès Texaco (1993-2018) (Buccina et al. 2013). Les tentatives de quantification des rejets de pollution font l'objet de différends entre les communautés locales et les opérateurs pétroliers. Ces rejets peuvent potentiellement contaminer les eaux souterraines utilisées par les communautés locales (Barraza et al. 2018; Wasserstrom, 2013).

Cette thèse est centrée sur la période 2001 à 2012. C'est la période la plus récente qui a suivi l'établissement d'une réglementation des hydrocarbures ; dont les données concernant cette période sont de meilleure qualité et donc plus fiables à utiliser pour les évaluations du risque, selon le Secrétaire National à la Planification et au Développement (SENPLADES, 2013).

1.2.4. Justification de la thèse et questions de recherche

Au moins deux tiers des données utilisées dans cette thèse ont été compilées à partir de données publiques en ligne, ou provenant du Conseil national des hydrocarbures et de l'accord de collaboration de recherche ANR-MONOIL et MAE-PRAS. Deux enjeux majeurs ont été sélectionnés comme études de cas pour l'évaluation de la vulnérabilité environnementale: (1) le patrimoine naturel et la biodiversité, et (2) les ressources en eaux souterraines. Ce choix a été fait en fonction de la pertinence de ces enjeux en général (Al-Adamat et al., 2003; Gardner et al., 2009; Sala et al., 2000; Shrestha et al., 2017) et pour la zone étudiée (Barraza et al., 2018; Barbieri et al., 2006; Provincial GAD, 2011, 2013; SENPLADES, 2013; Wernersson, 2004), mais a également été guidé par des critères pratiques : le temps, les contraintes budgétaires, ainsi que la disponibilité, l'accessibilité, l'exhaustivité perçue et la qualité des données.

Cette thèse s'inscrit dans le projet transdisciplinaire ANR-MONOIL, en partenariat avec les institutions équatoriennes (MAE-PRAS, 2016; MONOIL, 2017) et le Secrétariat national à l'éducation, à la science et à la technologie de l'Équateur (SENESCYT). Cette thèse vise à répondre à trois questions principales :

1. Quelles sont les variations spatiales et temporelles des émissions dangereuses quantifiées à la source ? À cette fin, (i) les déversements accidentels de pétrole brut, (ii) les rejets de gaz brûlés et (iii) les hydrocarbures stockés dans des fosses non revêtues ont été estimés.

2. Comment cartographier les niveaux actuels de vulnérabilité environnementale basée sur (i) les eaux souterraines et (ii) le patrimoine naturel, incluant la biodiversité, pour appréhender leurs variations spatiales ?

3. Comment pouvons-nous mettre en œuvre l'ERE en utilisant des estimations quantitatives des aléas liés aux hydrocarbures et de la vulnérabilité environnementale, combinées grâce à des méthodes indicielles et de superposition spatiale ? Ceci a été illustré avec le risque que représentent les fosses non revêtues pour les eaux souterraines.

En outre, un problème sous-jacent est abordé tout au long de la thèse : Dans quelle mesure les données publiques institutionnelles sont-elles adaptées à l'évaluation du risque dans la NEA?

Pour répondre à ces différentes questions, il a fallu dans un premier temps évaluer les aléas liés aux infrastructures pétrolières. Les déversements accidentels de pétrole, les fosses de stockage de résidus d'hydrocarbures non revêtues et les torchères utilisées pour le brûlage des gaz associés au pétrole ont été pris en compte. Les aléas ont été évalués par l’estimation des émissions de polluants à la source, c’est-à-dire au niveau de chaque infrastructure pétrolière pouvant potentiellement libérer des polluants de manière accidentelle ou intentionnelle. Les fréquences et les quantités de ces émissions n'avaient pas encore été estimées dans la NEA. Ces informations étaient nécessaires avant toute évaluation des aléas ou de l'exposition. La thèse a été divisée en quatre chapitres :

Le **chapitre I**, intitulé « Spatial analysis of accidental oil spills using heterogeneous data: a case study from the North-Eastern Ecuadorian Amazon » est une publication en révision destinée à évaluer les aléas liés aux déversements accidentels de pétrole ; plusieurs jeux de données ont été désagrégés et les données ont été associées aux infrastructures pétrolières sujettes à des déversements susceptibles de contaminer les sols et les eaux, en tenant compte notamment de la localisation spatiale de ces infrastructures au sein des blocs pétroliers, ce qui a permis de mettre en évidence la qualité hétérogène des données accessibles au public.

Le **chapitre II,** intitulé « Spatial inventory of selected atmospheric emissions from oil industry in Ecuadorian Amazon: insights from comparisons among satellite and institutional datasets », est une publication soumise qui traite des aléas liés aux torchères. Ces infrastructures sont perçues comme représentant un aléa important en raison du rejet constant dans l'atmosphère de gaz et d'aérosols, notamment sous forme de particules de noir de carbone.

Le **chapitre III** est consacré à la vulnérabilité environnementale, en particulier l’enjeu important que représente la biodiversité et, plus généralement, le patrimoine naturel. La problématique de l’évaluation, de la notation et de l’indexation spatiale de la vulnérabilité de cet enjeu est appréhendée et discutée. Son titre est « Spatial vulnerability assessment of natural heritage and biodiversity using current land use cover and nature protection levels ».

Le **chapitre IV** est une publication en préparation qui aborde le dernier composant d’une ERE, la combinaison des enjeux vulnérables indexés et des aléas liés à la production de pétrole brut, pour fournir des cartes de risque. De nombreuses combinaisons peuvent être évaluées mais, du fait de la limitation en temps, l'évaluation du risque a été illustrée en se basant sur un autre enjeu important : les eaux souterraines. Leur vulnérabilité a été évaluée, pondérée et indexée spatialement, puis combinée à la distribution spatiale des fosses de stockage non revêtues qui apparaissent comme la source de contamination la plus directe des eaux souterraines. Le titre de ce dernier chapitre est « Risk assessment of unlined oil pits to groundwater in Ecuadorian Amazon: A modified GIS-DRASTIC approach ».

Enfin, la thèse se poursuit par une discussion générale, qui précède une conclusion et des perspectives pour des recherches futures.

1. General Introduction

1.1 Risk associated to oil and gas activities

1.1.1. The concept of risk

The domain of risk has roots as ancient as human behaviour itself. The first thoughts about risk links to the fact that uncertainty of events occurring in the future was solely left to God's will (Rausand, 2011). There are unmeasurable and more measurable risks (Adger, 2006). The first may be envisioned with qualitative analysis while the latter can be dealt with some metrics allowing a quantification of these risks (SRA, 2015). Both intend to provide insights for proper decision making on a future course of action. Risks can be natural, ecological, social, political, technological, financial (Gleyze, 2002). An example of natural risk is the potential occurrence of a flood in a lower plateau, like 1931 flood in China, where nearly four million people were killed. The Chernobyl 1986 nuclear disaster was the materialization of a risk unperceived with catastrophic environmental consequences persisting nowadays. The global financial crisis of 2008 left millions of debt for the taxpaying citizens of affected countries due to lack of preventive measures, a crisis which did not reduce the risk of future economic collapse. These risks cannot be completely avoided, yet humans have improved technical ways to deal with them by reducing their chances to occur and by mitigating their consequences when they arise. Standard mitigation practices use bowtie analysis by implementing physical barriers at the hazard source (Shahriar et al., 2012), quantitative risk assessment (Mearns and Flin, 1995) and hazard identification studies or what-if analysis (Pinheiro, 2008).

1.1.2. Risk Assessment

Risk assessments are usually undertaken because of this need to understand the chances of something going wrong and try to avoid it, or to reduce its consequences. Yet, how to interpret and use them is not always clear. A widely operational method used in risk assessment is to define risk as a function of **hazard** and **vulnerability** as portrayed in Equation 1 (Jörn Birkmann and Wisner, 2006; Gleyze, 2002; Kazakis and Voudouris, 2015; Metzger et al., 2008).

A hazard is "the potential damage of a physical event, phenomenon or human activity that may cause the loss of life or injury, property damage, social and economic disruption or environmental degradation" (United Nations, 2004). A hazard is certain influence that may negatively distress a valued attribute of a system. It can be measured or approached in a quantifiable manner, by its frequency, intensity or its probability of occurrence (Gleyze, 2002). A hazard may be considered external to the system under evaluation, i.e., the flooding of 1931, the nuclear accident of Chernobyl. Nonetheless, some authors have contested concepts. Some studies consider internal hazards, i.e., a community may be threatened by unsustainable land use practices (Füssel, 2007). On the other hand, the unsustainable land use practice could also be perceived as internal deficiency of what makes the system vulnerable or susceptible to external hazards, e.g., erosion threatening soil fertility (Bayramin et al., 2008).

Vulnerability is the degree to which a hazard can harm the conditions of an attribute of concern or asset, depending on its intrinsic and specific properties (Berry et al., 2006). The asset that might be lost is considered as important by the society that benefits from its existence (Metzger et al., 2008). Examples of assets may be social and/or ecological. Ecological assets can be biodiversity, natural environment and its ecosystem services (Barraza et al., 2018; Castelo, 2017; Füssel, 2007; Metzger et al., 2006; Rahman, 2008). There are also social assets such as cultural identity of a community, patrimony susceptible to be affected by a threatening event or condition (Geyze, 2002). Assets can also be economic: income-generating activities, goods, capabilities. They can be a mixture of them such as agriculture (Berry et al., 2006; Luers et al., 2003). Hence, hazard and vulnerability could be analysed first as separated components (Birkmann, 2011) for a better understanding, although these elements are never fully independent, as one may consider that there are as many vulnerabilities as threatening hazards.

The term “vulnerability” is a concept whose use has increased in ecology issues for the past 20 years, with more than 170k publications appearing in the Web of science. It was previously used within evaluation on climate change impacts (Füssel, 2007; Kasperson and Kasperson, 2012; Smit and Wandel, 2006). Vulnerability is frequently associated to different types of hazards; indeed, studies often refer as vulnerability to flooding (Roy and Blaschke, 2015), climate change (Füssel, 2007; Kasperson et al., 2003; Preston et al., 2011), fires (Aretano et al., 2015), food insecurity (Bohle et al., 1994), water shortages (Becerra et al., 2016), etc. Several authors have provided conceptual background to vulnerability. Metzger et al. (2006), in a climate change specific context, define it as the degree to which an ecosystem is sensitive to a specific hazard, in this case global warming, added to the degree to which the society sector relying on the ecosystem services is unable to adapt to these changes (Metzger et al., 2006). Gleyze (2002) defines it in terms of the degree of actual or potential damage of an exposed element, system or subsystem, influenced by an event, stressor or hazard of a given intensity (Gleyze, 2002; Turner et al., 2003). In these aforementioned cases, the term refers to **specific**

vulnerability, e.g., a given chemical contaminant will potentially have a major impact under specific land uses or physicochemical context, while other contaminants will not (Beroya-Eitner, 2016; Stigter et al., 2006). Otherwise, if vulnerability is measured independently from the hazard, it is named **sensitivity** or **intrinsic vulnerability**. It refers to the intrinsic characteristics of an asset that render it sensible to degrade (Metzger et al., 2008). For instance, some factors responsible for intrinsic vulnerability are hydrological characteristics of an aquifer, the limited dispersal range of certain endemic species, or the self-organisation of local institutions.

One of the first concepts defined for environmental vulnerability was proposed within the ecotoxicological domain (Van Straalen, 1993), defined as the susceptibility of the natural environment to a negative impact of hazardous events or processes, i.e., series of chained events (Flax et al., 2002) or "the quantifiable degree or level of the foreseen damaging consequences of an event of a given intensity, natural, technological or industrial that is inflicted upon an asset" (Cutter, 1996; Gleyze, 2002; Miller et al., 2010). In this context, vulnerability may encompass three components: exposure, effect (also called intrinsic susceptibility or potential impact) and resilience (also named capacity to recover or adaptive capacity) (Beroya-Eitner, 2016; De Lange et al., 2010; Metzger et al., 2006; Van Straalen, 1993). Exposure is defined by Adger, (2006) as the nature and degree to which a system experiences environmental or socio-political stress. Therefore, clearly this reveals the overlapping of misleading concepts with vulnerability. Exposure has been occasionally considered not as part of vulnerability but as an independent condition of the system, usually in intrinsic vulnerability analysis. Besides, several authors coincide that exposure depends on the area of extent of the hazard (Adger, 2006; Stigter et al., 2006); and also will depend on the system or subsystem considered. For instance, a swamp ecosystem will not consider flood as a hazard but rather as an internal characteristic. On the other hand, flooding could be a hazard for human settlements. In other words, what for swampy ecosystems is a vital process, might be a threat for humans. Besides, variations in the vulnerability concept seem to be related to its application at different dimension scales. The assessment can reach environmental, social and institutional dimensions as portrayed by Birkmann (2011) and illustrated in Figure 1.

Once the system is vulnerable, the equilibrium or desired state could be changed towards a less productive or functional one. This capacity to return to the equilibrium state is coined under the term of "resilience", meaning the capacity of recovering functionality after a disturbance[1] has occurred (Walker et al., 2002). After such a distressful event the system can adopt measures or different behaviours to adapt to an alternate state i.e., adaptive capacity. An example of adaptive capacity is when human communities are not exposed to local water contamination, and thus not vulnerable, because a potable water system was built and brings water from distant non polluted sources (Saqalli et al., 2015). Resilience and/or adaptive capacity can be attained and improved with institutional re-structuration and land use planning (United Nations, 2004). Although being two different concepts, in this thesis a choice has been made to include resilience within capacity because this thesis will deal with information that might provide insights from maps aiming to plan for adaptation or resilience in the future by potentially reorganizing the territory's land use.

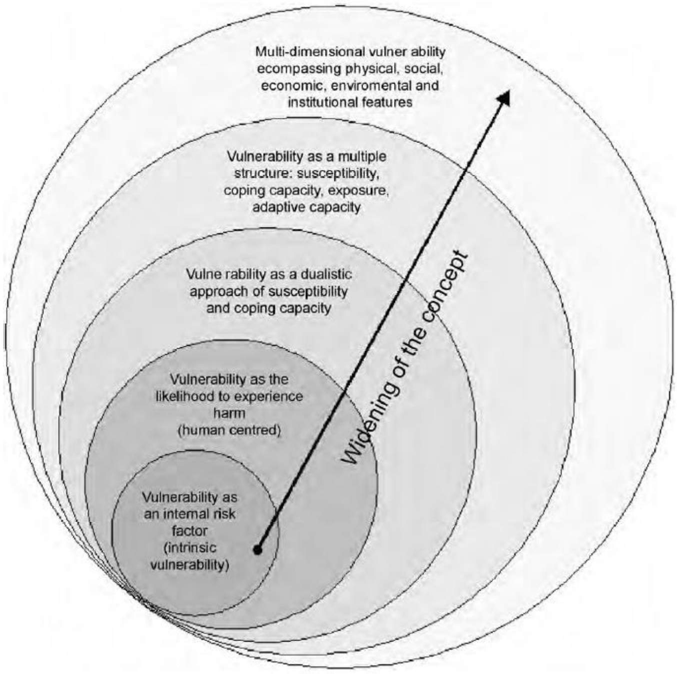


Figure 1. Multiple dimension scales of the vulnerability concept (Birkmann, 2011).

In this thesis, a risk assessment for environmental assets in a context of crude oil activities is proposed. The methodological approach implemented for **Environmental risk assessment (ERA)** is explained with a flowchart illustrated in Figure 2. In spite of the several possible approaches to evaluate risk, in this thesis risk assessment was based on a spatial explicit and operational methodology combining hazard and vulnerability, whose definitions are comprised in Table 1. The method can be implemented at the watershed, landscape, regional and global

[1] A hazardous process or event that pressures the system to an alternative state.

scales. The approach is a coupled **quantitative** and **spatial overlay-index method** summarized in the Equation:

$$Risk = \sum_{i=1}^{n}(Hp_i \times Hw_i) \times \sum_{j=1}^{m}(Vp_j \times Vw_j) \quad \text{(Eq.1)}$$

Where:

n: number of parameters constitutive of the hazard

Hp_i: value of parameter *i* characterizing the hazard

Hw_i: weight given to parameter *i* in hazard assessment

m: number of parameters constitutive of the vulnerability

Vp_j: value of parameter *j* characterizing the vulnerability

Vw_j: weight given to parameter *j* in vulnerability assessment

Table 1. Definitions of the main concepts used in this study.

Hazard (*H)*	Adverse event, characterized in terms of probability, frequency, intensity and potential consequences (Gleyze, 2002). This thesis will focus on human-related events.	
Vulnerability (*V*)	Characteristic of an asset related to the potential degree of harm required to change its conditions when exposed to hazards (Adger, 2006; Metzger et al., 2008)	Sensitivity is defined as intrinsic characteristics of an asset that render it sensible to degrade. (Metzger et al., 2008)
		Resilience is the capacity of restoring the system functionality after a disturbance has occurred (Smit and Wandel, 2006; Walker et al., 2002).

Both hazard and vulnerability can be defined using several constituent parameters, which can be a quantity or a qualitative value that can be classified. As indicated Equation 1, each of these parameters can be weighted, to quantify its importance in the overall hazard or vulnerability value. If this importance is unknown, weighting can be ignored or considered uniform for all parameters (Cutter, Mitchell and Scott, 2000).

The factors influencing the resilience of the system can be taken into account in Equation 1, as parameters constitutive of vulnerability. In contrast, it is not necessary to explicitly refer to exposure, as this is taken into account implicitly: risk resulting from the multiplication of hazard and vulnerability, it vanishes when one of these terms equals zero. In other words, in a spatial approach, risk is absent in areas where there is no overlap between hazard and vulnerability.

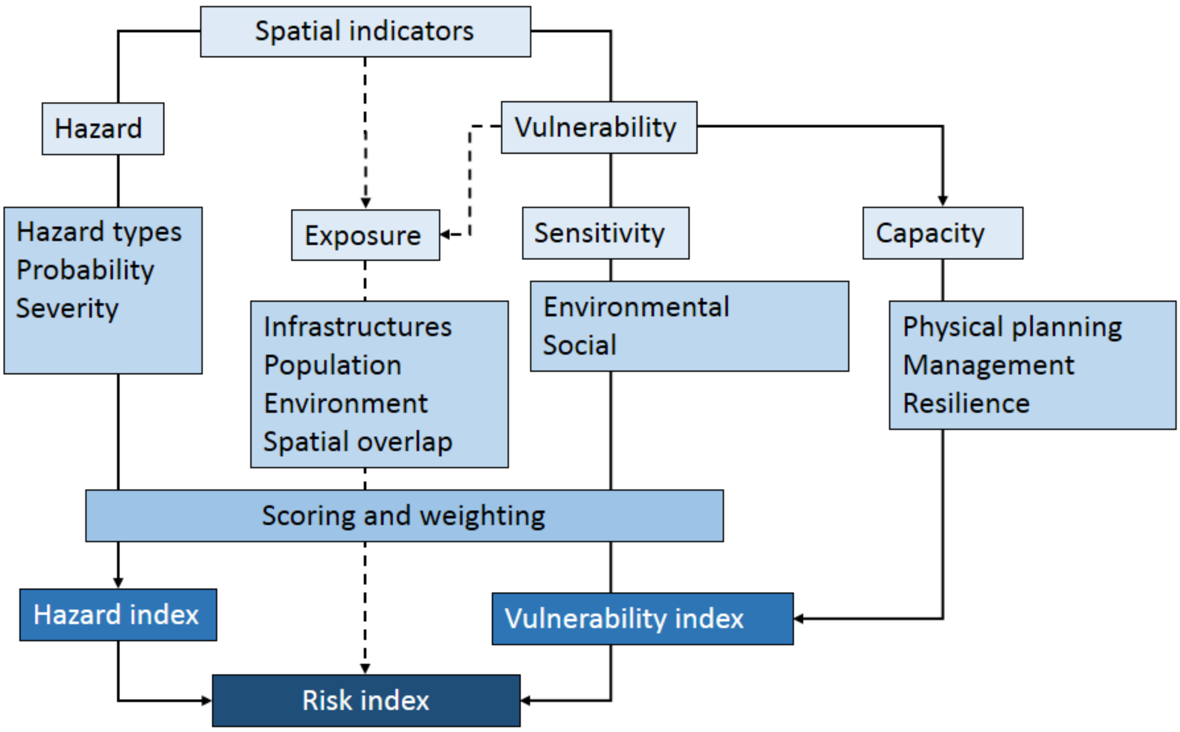


Figure 2. Flowchart of the general methodological approach. Modified from the CHARIM project (2005). The exposure component, since not explicitly considered in this thesis, has been displayed with dotted lines, to represent its importance in vulnerability assessment but its implicit consideration as part of the overlaying of maps during risk assessment in this spatial approach.

1.1.3. Environmental risk assessment for crude oil activities

The case of crude oil is important because it is the most traded commodity in the world. It increases revenues and raises living standards of export countries (Acosta, 2006). During the

upstream[2] and midstream[3] crude oil production chain, several products resulting from such activities may (accidentally or intentionally) be discharged to the proximate environment (Jernelöv, 2010). They may become hazardous, disturb, stress or kill plants and animals, and contaminate soils and waters. Figure 3 illustrates the production chain with regard to the various locations where polluting emissions are most likely to occur.

Upstream oil and gas production raises risk from hazardous emissions attributed to several routine operations, including gas extraction and accidental oil spills (Jernelöv, 2010). All these operations are specifically and spatially allocated to different infrastructures. Oil spills involve pipelines, oil wells[4] / platforms[5] or oil separation batteries[6], and result from both regular operations and severe accidents (Burgherr, 2007). Accidental oil spills can impact soil and water, thus local population health and agricultural production (Chang et al., 2014). Oil related toxic chemical contaminants, such as Polycyclic Aromatic Hydrocarbons (PAH) and heavy metals, can leach into soils, reach surface drinking water and groundwater, and degrade vulnerable ecosystems (Bondur, 2011; Wernersson, 2004).

Venting and flaring produce atmospheric pollutants, which comprise aerosol particulate matter (PM), including black carbon (BC), and greenhouse gases (GHG) such as methane, carbon dioxide or nitric oxide (NO_3). Gas flaring has the potential to acidify rainwater when sulphide hydrogen (H_2S) is released (Anejionu et al., 2015b). Black carbon (BC) is released during gas flaring and acts at a local level, resulting in soil calcination, degradation and destruction of vegetation (Solov, 2011). BC is a major light absorbing fraction of PM aerosols (Bond and Bergstrom, 2006) formed from incomplete combustion. Its radiative heating properties (Jacobson, 2001) transform it into a short-lived climate forcer. Since it is an important constituent of fine particles and serves as carrier PAHs (Janssen et al., 2011), BC also has adverse effects on health (McEwen and Johnson, 2012), by increasing the risk of cardiopulmonary morbidity and mortality (Anejionu et al., 2015a; Giwa et al., 2014; Huang and Fu, 2016; Janssen et al., 2011). BC is classified as carcinogenic to humans by the International Agency for Research on Cancer (IARC, 2012). On the other hand, carbon dioxide

[2] Refers to oil exploration and extraction activities. In this study exploration is not considered, only extraction is considered.
[3] Refers to transportation, separation of oil, water and gas at separation batteries and refining.
[4] A producing well with oil as its primary commercial product. Oil wells almost always produce some gas and frequently produce water. Most oil wells eventually produce mostly gas or water.
[5] Space dedicated to barren land where oil production infrastructure, like oil wells, separation batteries and other related equipment is placed.
[6] A cylindrical or spherical vessel used to separate oil, gas and water from the total fluid stream produced by a well.

(CO_2) and methane (CH_4) are climate change forcers with significant warming potentials (Soltanieh et al., 2016) and methane has energy generation capabilities. These capabilities are wasted off by venting or flaring because of security and safety reasons (Anenberg et al., 2012; Boden et al., 2012; Dlugokencky, 2003; Robalino-López et al., 2014; Simpson et al., 2012).

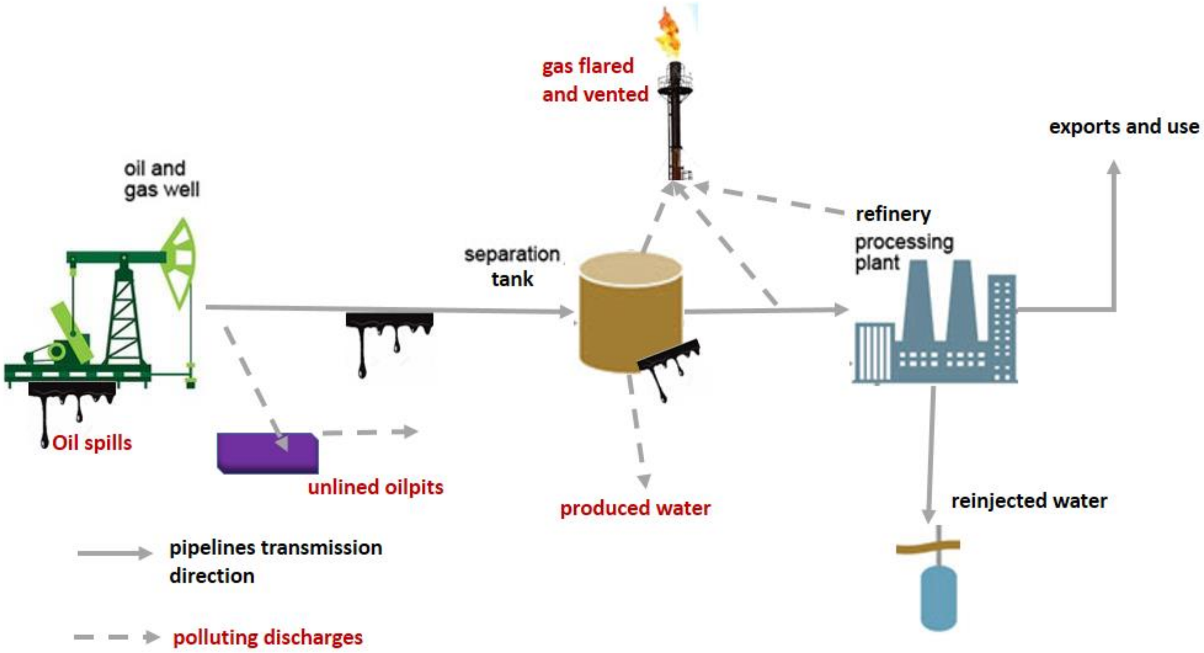


Figure 3. Upstream and midstream crude oil production chain. Oil is produced by extracting techniques at oil wells, then muds and residues are collected in pits, the oil and gas are transported via pipelines to separation batteries where oil, gas and produced water are separated. Crude oil is transported to refineries and to export ports. This process is not exempt of accidental or intentional discharges that can reach environmental compartments.

Another relevant contaminant source is related to pits. The purpose of a pit is to hold drilling cuts and drilling mud which contain heavy metals, and total petroleum hydrocarbons (TPH) that are not collected by the pipeline (MAE-PRAS, 2016). TPH is a term used to describe a large family of several hundred chemical compounds that originally come from crude oil (ATSDR, 2011a) including PAHs (ATSDR, 2011b). In addition, pits retain significant amounts of produced water which is pumped to the surface alongside oil and gas, precipitating Naturally Occurring Radioactive Materials (NORMs) from drill cuttings depending on geological formation. Examples of these are: uranium, thorium and high amounts of radium (^{226}Ra and ^{228}Ra), co-precipitated with barium salts ($BaSO_4$) which are somewhat soluble in water and therefore mobile (Bakke et al., 2013; Jerez Vegueria et al., 2002). Pit construction standards require particular volumetric dimensions, i.e., size should be sufficient to ensure adequate storage until closure, taking into account historical rainfall patterns; depth should be such that the bottom does not penetrate groundwater, or such that the pit contents do not adversely impact groundwater or surface water. A site chosen for implantation of oil pits should be investigated in terms of presence of aquifers, and a specific action plan should be elaborated to protect them (EPA, 2014).

Polluting emissions related to oil activities, such as spills of crude oil, brine and mine residues (releasing mainly hydrocarbons and heavy metals), or the burning of gases releasing aerosols in the form of particles, including black carbon, can have **ecotoxicological effects**. The ECOTOX database from the Environmental Protection Agency (EPA) provides 3,316 search results for all the chemical compounds resulting from oil activities. Table 2, 3 and 4 synthesize some of the most notable effects occurring in natural conditions influenced by the pollutants considered. The **toxicological effects** of these pollutants to humans are provided in these tables as well. The intentional or accidental release of crude oil related contaminants resulted in long-term pollution of air, soil, surface waters and groundwater in some regions (Barraza et al., 2017; Wernersson, 2004). For instance, cacao plantations in the NEA have been reported to have high concentrations of cadmium which in part coming from anthropogenic activities[7] (Barraza et al., 2017) and TPH proved to be toxic for certain aquatic animals in Rio Rumiyacu, Dayuma, Ecuador (Wernersson, 2004).

Alternative applicable methods regarding risk assessments are numerous. The main currently used can be categorized as administrative, academic and compartment thresholds.

[7] Cd is a potential cancerigenous heavy metal produced from oil and gas activities, land use practices and also from natural volcanic activity in the NEA.

Table 2. Environmental and health effects of main non-metallic crude oil substances, mostly comprised within TPH (GF = gas flaring; OS = oil spills; OP = unlined oil pits).

Compounds Fractions	Hazard source	Human health effects	Ecotoxicological and environmental effects	Reference
Aliphatic lighter BTEX C6-C15	GF, OS	Carcinogenic to humans (benzene and benzoαpyrene) Leukaemia Decreased foetal body weight Skin irritation, Central nervous system depressant Destroying membrane of red blood cells	Increased kidney and adrenal weight Reduce nerve conduction velocity Decreased lymphocyte count Skin irritations	(IARC, 2012) (Soltanieh et al., 2016) (Vanos et al., 2013) (Argo, 2001)
Aliphatic heavier C16-C35	GF, OP, OS	Affects central nervous system Decrease immune function Blood transport deficiency	Death to *Echinocloa crassipes* and *E.ploystachya* Reduce leaf width on plants Reduction of total sugar on crops Reduced chlorophyll content and internode length for *Eupatorium odoratum*	(Lopes et al., 2009) (Dung et al., 2008) (Singh and Chandra, 2014) (Isichei and Sanford, 1976)
PAHs	GF, OP, OS	Skin irritation Decrease immune function Cataracts, Kidney and liver damage Carcinogenicity and mutagenicity	Weight reduction and genotoxicity in *Daphnia magna* Death to *Hyalella azteca* Calf stillbirth and death Reduced chlorophyll content in plants *Melastomataceae, Fabaceae, Rubiaceae, Euphorbiaceae* Mutagenicity in *Salmonella typhimurium* Death to *Chironomus kieensis* Skin cancer in mouse and hamster	(Abdel-Shafy and Mansour, 2016) (Wake, 2005; Wernersson, 2004) (Waldner, 2008; Waldner et al., 2001) (San Sebastián et al., 2001) (Arellano et al., 2017) (ATSDR, 2011b) (IARC, 2012) (Lacerda et al., 2014)

Table 3. Gas flaring pollutants such as greenhouse gases and black carbon according to their environmental and human health effects.

Compounds	Health effects	Reference	Ecotoxicological and environmental effects	Reference
NOx, VOC's	Kidney disease Blood disorder Bronchitis Stomach ulcers	(Egwurugwu et al., 2013) (Soltanieh et al., 2016)	Soil recalcitration[8] Acidic rain	(Solov, 2011) (Anejionu et al., 2015b) (Ite and Ibok, 2013)
CO_2	Non-toxic	(IPCC, 2013)	Reference Global warming potential (GWP) Climate change precursor	(IPCC, 2013)
CH_4	Non-toxic	(IPCC, 2013)	28 times higher in GWP than CO_2 in 100 years. Climate change precursor	(IPCC, 2013)
BC	Cancerigenous in humans (Lung cancer) Cardiopulmonary disease Asthma, pneucomoniosis	(Janssen et al., 2011)	Short-term high radiative forcing	(IARC, 2012) (Jacobson, 2001) (Soltanieh et al., 2016)
H_2S	Not listed – not acutely toxic	(IARC, 2012)	Death to fish *Piaractus brachypomus* and *Piaractus brachypomus* Respiratory, neurological and olfactory effects Medium sensitivity for *Daphnia magna*	(Reátegui-Zirena et al., 2012) (Küster et al., 2005)

[8] Under temperature inversion effects.

Table 4. Metals found in crude oil, mostly disseminated via oil spills and unlined pits, and their carcinogenic and ecotoxicological effects.

Compounds	Hazard source	Carcinogenic effects - IARC[9]	Ecotoxicological effects [10]- EPA-ECOTOX[11]	Source
Hg	OS, OP	Non carcinogenic	Higher body weight and growth in *Hoplias malabaricus* Structural changes on larvae in *Hoplias malabaricus*	(Webb et al., 2015)
Ni	OS,OP	Carcinogenic to humans	Reduce overall weight in *Chironomus dilutus* Bioaccumulation in worms in *Hyallela azteca* Reduce number of progeny counts in *Cerioduphniu dubiu*	(Erickson et al., 2011) (Liber et al., 2011) (Spehar and Fiandt, 1986)
V (V_2O_5)	OS,OP	Possible carcinogenic to humans	Stimulates algae grow in *Melosira. Varians*	(Patrick et al., 1975)
Ti (TiO_2)	OP	Possible carcinogenic to humans	Not listed Fibrosarcomas and Limphosarcomas in rats	 (Nordberg, 2007)
As	OP	Carcinogenic to humans	Anomaly weight in spleen of fish Histological changes in fish Reduce overall weight in worms Bioaccumulation in worms	(Erickson et al., 2011; Liber et al., 2011) (Liber et al., 2011)
Pb	OP	Possible carcinogenic to humans	Population growth in algae	(Vighi, 1981)
Cd	OP	Carcinogenic to humans	Bioaccumulation in amphibians Bioaccumulation in flowers of plants Reduced photosynthetic activity	(Nebeker et al., 1995) (Reinke, 2011) (Selby et al., 1985)
Cr (VI)	OP	Carcinogenic to humans	Decrease in blood clotting time in *Tilapia spaarmani* Decrease in cell viability in *Pimephales pomelas* and *Carassius auratus*	(Velma et al., 2009)

[9] International Agency for the Research on Cancer. World Health Organization.

[10] Typical test species experiencing mortality for all trace metals are: *Hyallela azteca,Lumbriculu variegatus, Pimephales promelas, Salvelinus fontinali*s, *Oncorhynchus mykiss, Jordanella flroidus, Chrionomus dilutes, Lepomis macrochirus, Salvelinus namaycus, Simocephalus serrulatus, Ceriodaphnia reticulate, Cyprinus carpio.* Selection of mortality effects for <24 hours in aquatic and natural environment species by exposure due to leaching and flow through.

Table 4. (continued) Metals found in crude oil, mostly disseminated via oil spills and unlined pits, and their carcinogenic and ecotoxicological effects (continued).

Compounds	Hazard source	Carcinogenic effects - IARC[12]	Ecotoxicological effects [13]- EPA-ECOTOX[14]	Source
Ra	OP	Carcinogenic to humans[15]	Endocrine disruption on *Oncorhynchus mykiss*	(Bakke et al., 2013)
$BaSO_4$	OP	Not listed	Nephropathy in male mice Increased kidney weight in female mice	(ATSDR, 2019)

[12] International Agency for the Research on Cancer. World Health Organization.

[13] Typical test species experiencing mortality for all trace metals are: *Hyallela azteca,Lumbriculu variegatus, Pimephales promelas, Salvelinus fontinali*s, *Oncorhynchus mykiss, Jordanella flroidus, Chrionomus dilutes, Lepomis macrochirus, Salvelinus namaycus, Simocephalus serrulatus, Ceriodaphnia reticulate, Cyprinus carpio.* Selection of mortality effects for <24 hours in aquatic and natural environment species by exposure due to leaching and flow through.

[15] NORMs = Naturally Occurring Radioactive Material.

1.2. The North-eastern Ecuadorian Amazon as a case study

1.2.1. General overview and importance of the study area

Ecuador is the fifth oil producer in Latin America (OECD, 2016). Ecuador has a five decades history of crude oil production, concentrated in the North-eastern Ecuadorian Amazon (NEA). However, the NEA is one of the most important biodiversity hotspot in the world (Bass et al., 2010): The NEA includes flora and fauna species representing high biodiversity even within Amazon basin intra-comparisons (Bass et al., 2010; Myers et al., 2000). At least, 210 mammals, 131 amphibians, 558 birds and more than 3,000 vascular plants species have been reported only in Yasuní National Park (Bass et al., 2010; Finer et al., 2008). Biodiversity is also important for the development of medicines (Chaudhary et al., 2015; Neergheen-Bhujun et al., 2017). The NEA is also home of eight indigenous populations: Kichwa, Waorani, Siona, Secoya, Cofán, Shuar, Tagaeri and Taromenane, among which the last two remain in voluntary isolation from western civilization.

Historically, the NEA is an important environmental study case as it has drawn attention of many scientists for decades in the fields of socio-ecological systems (Mena, 2008; Messina and Walsh, 2005; Walsh et al., 2008b) and complexity theory (Walsh et al., 2008a). This is notably because it is a historical high rate frontier deforestation setting (Messina et al., 2006; Sierra, 2000; Walsh et al., 2008a), where immigration pressures have played an important role (Holland et al., 2014; C. Mena et al., 2006), partially because of conflicts in land use practices between inhabitants with extensive agricultural practices vs. populations who use smaller and subsistence agriculture (Holland et al., 2014; Mena et al., 2006; Messina et al., 2006; Pan et al., 2004). These pressures are the result of oil driven development, which has extensively influenced road opening and consequent land use changes (Baynard et al., 2013).

1.2.2. Biophysical characteristics

The study area (Figure 4A) was restricted to the provinces of Sucumbíos and Orellana in the NEA (~144-900 m a.s.l., Amazon lowlands), representing a 35,051 km^2 area (Figure 4B). This study excluded rivers with high flow rates (i.e., Napo, Tiputini, Coca, Payamino, Putumayo, Cuyabeno and Aguarico) where the surface waters and their interactions with groundwater are highly dynamic, therefore difficult or impossible to evaluate using scarce data.

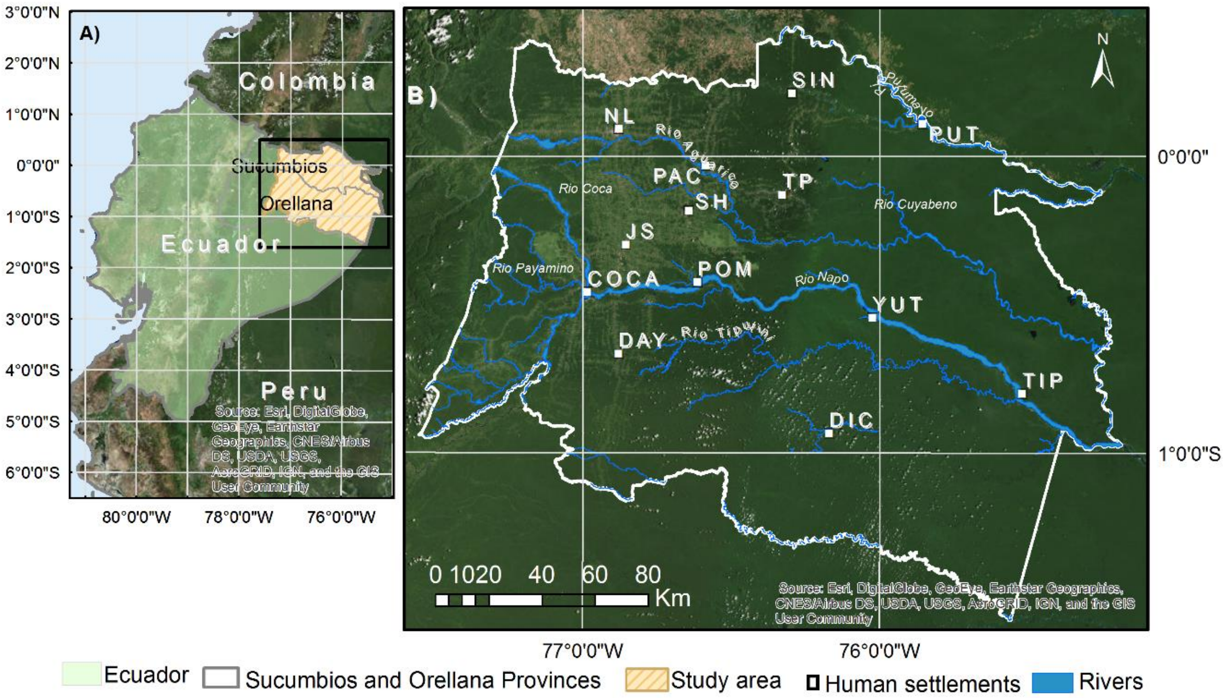


Figure 4. Location of the study area. (A) The North-eastern Ecuadorian Amazon (NEA) in Ecuador, bordering Colombia in the North and Peru in the South. (B) The study area within the two NEA provinces of Sucumbíos and Orellana including major rivers and using as background a land cover mosaic, courtesy of GeoEye, Digital Globe, (2018). Represented human settlements are abbreviated: DAY= Dayuma; NL = Nueva Loja (aka Lago Agrio); SH = Shushufindi; TP = Tarapoa; PAC = Pacayacu; POM = Pompeya; PUT = Putumayo; COCA = Puerto Francisco de Orellana; JS = Joya de Los Sachas; YUT = Yuturi; DI = Dícaro; SIN = Singue; TP = Tiputini.

The study area is characterized by a warm climate with an annual temperature range of 20°C to 30°C, and an average annual rainfall of 2,900 mm.yr^{-1} (Institute of Meteorology and Hydrology-INHAMI). The hydrology regime is irregular with 1,000 to 5,000 m^{3}.s^{-1} daily discharges, and characterized by flash floods, due to high sensitivity to rain events (Laraque et al., 2007). The Intertropical Convergence Zone (ITCZ) is responsible for complex atmospheric processes in the NEA, which is influenced by interchangeable wind direction throughout the year, i.e., north to southward direction from October to April and south to northward from May to September (Palermo and Parra, 2014).

1.2.3. Lithology and hydrogeology

Lithology refers to soil characteristics and properties. In the NEA region soils are mainly composed of three types (MAGAP-SIGTIERRAS and Tracasa-nipsa, 2015; Sourdat and Custode, 1982):

(i) Amazonian basin low plains in recent sedimentary rocks: well drained acidic montmorillonite and kaolinite clays of volcanic origin found over 600 m.a.s.l. These soils are low ancient rounded mountains formed by dissection of sedimentary banks of similar thickness and agglomerates, with slopes lower than 40%. Low fertility soils.

(ii) Amazonian basin low hilled modelled by ancient weathered sedimentary rock: heterogeneous drained sandy-silt mixed soils up to four meters over clay substrate of alluvial origin. These are most recent soils located lower than 600 m.a.s.l., with slopes lower than 10%.

(iii) Far Andean detritus material derived low-mountains: developed plains over sandy volcanic material between 300-900 m.a.s.l. clayed textures from the Tertiary, usually covered by forest or indigenous crop lands with low fertility.

Hydrogeological settings and properties indicate geological formations date from 5 to 56 million years ago (from the Eocene epoch to the Quaternary period). The NEA encompasses seven geological formations, where three possess pervious characteristics (Burbano et al., 2015):

(i) High permeability in alluvial lower plains containing sands and fluvial sediments.

(ii) Medium permeability in lime, sands pyroclastic lahars conglomerate formations.

(iii) Low permeability in mudstones, lime silty tidalites formations.

Thickness range from 150 to 548 meters (in locations where measures are available). Further non-exhaustive information on age, drainage directions and media composition of geological formation are detailed in Annex and Figure C.4.

1.2.4. Nature conservation measures

Probably it is not possible to explain the conservation measures of natural heritage implemented in the NEA without explaining the colonization process propelled by oil reserves discovered underneath the Amazonian forest. Two agrarian reforms, in 1964 and 1973, encouraged the colonization of forested lands in the NEA, as colonists engaged in forest clearing ensured title to lands (Holland et al., 2014). As a reaction to environmental degradation, several natural heritage sites have been declared by central government, Autonomous Local Governments (GADs), or have been community based or private owned (Ambiente, 2018). These conservation zones intended to regulate the socio-economic activities within geographical boundaries, notably via the protection of several environmental assets in the form of protected areas (PA). In 1979, Cuyabeno Fauna Reserve and Yasuní National Park, the largest Amazonian reserves most biologically and culturally diverse, were created to protect natural heritage (Ambiente, 2018). In 1981, the Ecuadorian Institute for Agrarian Reform and Colonization (IERAC) implemented a conservationist law named the Forestry Law. This law set the grounds for the network of natural heritage sites in Ecuador. In 1985, the wetland reserve of Limoncocha, fostering a lagoon with highly endemic aquatic fauna and flora, was created (Ambiente, 2018). According to Messina et al. (2006), the protection level of nature heritage has been successful at avoiding further deforestation in these areas (Figure 5).

However, the reshape of boundaries to allow oil production fields within several PA resulted in the establishment of the *Patrimony Forest* buffer zones. These zones limited but at less extent, the extractive industries, notably oil production (Mena et al., 2006). Patrimony Forest is an administrative category of lands where extractive activities are restricted and where land is communized, i.e., land titles are collective, impossible to be sold and managed by a collective entity; land usufruct rights are individual, can be transmitted and inherited but not sold (Mena et al., 2006). In addition, the GAD, are administrative divisions of the territory. GADs have jurisdictions at the local (parish[16]) and regional (province[17]) levels. GADs are entities that have certain autonomy to decide over the land use planning, have created several *Protected Forests* supported by international efforts to establish Biosphere Reserves, a heritage status awarded by the United Nations Educational, Scientific and Cultural Organization (UNESCO). Last but not least, the national initiative of the *Socio-Bosque Program*[18] (2008) should also be cited. This initiative's objective is to provide some monetary value to communities or individuals entitled land owners. The monetary incentives are given per hectare that is conserved, restored or managed under sustainable land use practices. Private or communal forested lands are eligible under the condition of maintaining the forest for at least twenty years and being outside of the system of protected areas (PSB-MAE, 2013). Table 4 indicates the main conservation events which were prompted to be declared for nature protection from increasing socio-economic activities, having oil production as main driver.

[16] Third-level administrative units of Ecuador. The Cantons of Ecuador are divided into parishes which are similar to municipalities or communes in many countries. There are over 1,500 parishes in Ecuador.

[17] Provinces are the first administrative level

[18] Initiative from the Ministry of the Environment of Ecuador (MAE).

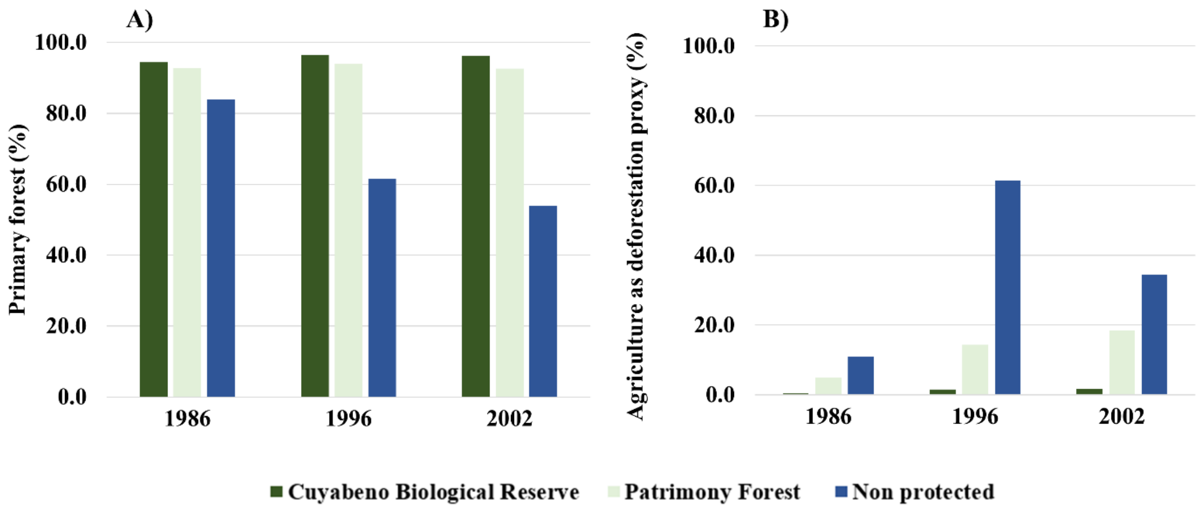


Figure 5. Temporal evolution of land use by main protection status in the province of Sucumbíos (part of NEA). Percentage of primary forest (PF) vs. agriculture (AG) as proxy of deforestation. A) Primary forest percentage remaining per status of protection. B) Agriculture extent by status of protection. Adapted from Messina et al. (2006).

1.2.5. Oil activities in the NEA

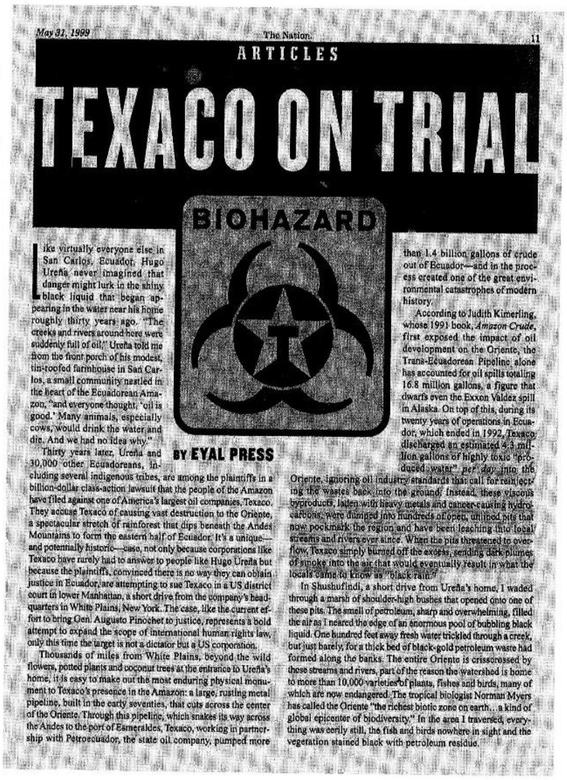
In Ecuador, crude oil was discovered in 1937, with the prospecting activities by the Royal Dutch Shell Company, yet in 1960 the American Texaco started prospecting in the NEA. Actual production started only in 1967 with the American company Texaco, locally named Texpet in Nueva Loja[19]. Later, the production started for the giant oilfields of Nueva Loja and Shushufindi, and then for the Auca field. Seven operators[20] were awarded contracts for the production of the NEA crude oil. In 1972 the state owned "Corporación Ecuatoriana de Petróleos del Ecuador" (CEPE), now PetroEcuador, was created. The first Trans-Ecuadorian crude oil pipeline (SOTE) connected the Amazon with the port of Esmeraldas, thus initiating the crude oil production famous period named the "oil boom of 1972-1982" (Acosta, 2006; Larrea, 2006). The revenues from crude oil exports doubled the per capita income and Ecuador became dependent on this primary commodity (Larrea, 2006). To summarize the long Ecuadorian crude oil history, three management time periods have been proposed, spanning

[19] Nueva Loja is also called Lago Agrio. It means Sour Lake which commemorates the city where oil production started in Texas, USA. The first oil well produced 358 t/day.

[20] Shell, Standard Oil, California Oil, Tennessee, and the Western Geophysical Co, Ecuadorian company (CEPE) and Texaco.

five decades of oil production (Juteau et al., 2014). The time period are labelled hereafter as T1 from 1972 to 1992, T2 from 1992 to 2000 and T3 from 2001 onwards (Table 4).

Besides the numerous economic benefits of crude oil exports, these activities resulted in accidental or intentional pollution discharges as explained in section 1.3. These extractive activities may damage the environment, as well as the exposed local communities, who have claimed not benefiting from these revenues (Buccina et al., 2013; San Sebastián and Hurtig, 2005; SENPLADES, 2013). This resulted in the infamous long-term (1993-2018) Texaco trial (Buccina et al., 2013) (Figure 6). Some attempts to quantify the pollution discharge have been a matter of disputes between local communities and oil operators. For instance, between 1972 and 1992, different numbers of discharged amounts have been reported. The Programme of Environmental and Social Remediation (PRAS) reported a total of 505.6 kt. of poured crude oil (MAE-PRAS, 2016). The Amazon Defense Front[21] (ADF) reported 6.6 Mm^3 gas flared and 155 Mt crude oil spilled (Amazon Defense Front, 2008). 54.7 kt. crude oil and 96.8 Mt of produced waters were indicated by others (Kimerling, 1990). During the Remedial Action Plan (RAP) in 1995, only 158 pits have been alleged to be merely covered with a barren soil coating. Until 2013, more than 2,000 oil pits were documented, but only for 1,420 oil pits were measured and reported to have highly variable volumes (MAE-PRAS, 2016). These discharges may potentially contaminate ground waters that are used by the local communities (Barraza et al., 2018; Wasserstrom, 2013).


May 31, 1999 The Nation. 11

ARTICLES

TEXACO ON TRIAL

BIOHAZARD

BY EYAL PRESS

Like virtually everyone else in San Carlos, Ecuador, Hugo Ureña never imagined that danger might lurk in the shiny black liquid that began appearing in the water near his home roughly thirty years ago. "The creeks and rivers around here were suddenly full of oil," Ureña told me from the front porch of his modest, tin-roofed farmhouse in San Carlos, a small community nestled in the heart of the Ecuadorean Amazon, "and everyone thought, 'oil is good.' Many animals, especially cows, would drink the water and die. And we had no idea why."

Thirty years later, Ureña and 30,000 other Ecuadoreans, including several indigenous tribes, are among the plaintiffs in a billion-dollar class-action lawsuit that the people of the Amazon have filed against one of America's largest oil companies, Texaco. They accuse Texaco of causing vast destruction to the Oriente, a spectacular stretch of rainforest that dips beneath the Andes Mountains to form the eastern half of Ecuador. It's a unique—and potentially historic—case, not only because corporations like Texaco have rarely had to answer to people like Hugo Ureña but because the plaintiffs, convinced there is no way they can obtain justice in Ecuador, are attempting to sue Texaco in a US district court in lower Manhattan, a short drive from the company's headquarters in White Plains, New York. The case, like the current effort to bring Gen. Augusto Pinochet to justice, represents a bold attempt to expand the scope of international human rights law, only this time the target is not a dictator but a US corporation.

Thousands of miles from White Plains, beyond the wild flowers, potted plants and coconut trees at the entrance to Ureña's home, it is easy to make out the most enduring physical monument to Texaco's presence in the Amazon: a large, rusting metal pipeline, built in the early seventies, that cuts across the center of the Oriente. Through this pipeline, which snakes its way across the Andes to the port of Esmeraldes, Texaco, working in partnership with Petroecuador, the state oil company, pumped more than 1.4 billion gallons of crude out of Ecuador—and in the process created one of the great environmental catastrophes of modern history.

According to Judith Kimerling, whose 1991 book, *Amazon Crude*, first exposed the impact of oil development on the Oriente, the Trans-Ecuadorean Pipeline alone has accounted for oil spills totaling 16.8 million gallons, a figure that dwarfs even the Exxon Valdez spill in Alaska. On top of this, during its twenty years of operations in Ecuador, which ended in 1992, Texaco discharged an estimated 4.3 million gallons of highly toxic "produced water" *per day* into the Oriente, ignoring oil industry standards that call for reinjecting the wastes back into the ground. Instead, these viscous byproducts, laden with heavy metals and cancer-causing hydrocarbons, were dumped into hundreds of open, unlined pits that now pockmark the region and have been leaching into local streams and rivers ever since. When the pits threatened to overflow, Texaco simply burned off the excess, sending dark plumes of smoke into the air that would eventually result in what the locals came to know as "black rain."

In Shushufindi, a short drive from Ureña's home, I waded through a marsh of shoulder-high bushes that opened onto one of these pits. The smell of petroleum, sharp and overwhelming, filled the air as I neared the edge of an enormous pool of bubbling black liquid. One hundred feet away fresh water trickled through a creek, but just barely, for a thick bed of black-gold petroleum waste had formed along the banks. The entire Oriente is crisscrossed by these streams and rivers, part of the reason the watershed is home to more than 10,000 varieties of plants, fishes and birds, many of which are now endangered. The tropical biologist Norman Myers has called the Oriente "the richest biotic zone on earth... a kind of global epicenter of biodiversity." In the area I traversed, everything was eerily still, the fish and birds nowhere in sight and the vegetation stained black with petroleum residue.

Figure 6. Press article on Texaco trial published in 1999 in the American Journal "The Nation", with phrases such as: "There can be little doubt Texaco was aware of its production methods in the Amazon departed from accepted standards".

A few years after departing of Texaco, the indigenous communities formed the "Union de Afectados de la Amazonia" (Amazon Defense Front or ADF, by its abbreviation in English), whose purpose was to legally pursue the international oil firm due to alleged environmental

[21] Non-governmental organisation established to claim for the alleged contamination left by Texaco-Chevron.

pollution and related diseases, i.e., skin irritations, nauseas and headaches, increased cancer incidence and increased mortality (San Sebastián et al., 2001; San Sebastián and Hurtig, 2005). The ADF engaged in a long legal prosecution that aimed to repair the damages caused in this pristine area, estimated at a cost of US$18 billion. During the process, Texaco Inc. and Chevron Corp. were merged. The case continued for twenty five years, until an international court in The Hague, ruled in favour of the Chevron-Texaco firm in September, 2018 (BBC News, 2018).

The case was always conflicting around issues of the respective responsibility of the state owned company Petro-Ecuador and the Chevron-Texaco company. In fact, several issues may arise from the different management practices, monitoring and implemented technology regarding the upstream production chain quality, according to either *state* or *private* operators. It is unclear how private operators' improvements are made, whereas state operators are more easily accounted, at least regarding oil spill prevention. Self-reported data is used in these claims and comparison between companies is very limited (Frynas, 2012).

T3 extends in practical terms, to the present day. More than 70% of oil production, corresponding to the majority of oil blocks, was assigned to Petro-Amazonas in 2018. Figure 7 shows main oil operators in the region as designated in 2018.

Table 5. The three main different periods recognized during the crude oil production in Ecuador, compared with main legal events and conservation measures. Table is adapted from Juteau et al., (2014) and Saqalli et al. (2018).

Time period	Oil activities and type of management	Conservation measures	Main legal actions and remediation plans
Before 1970	Petroleum prospecting by Texaco First settlements of *colonist* families, brought by military plane.	-	Law of the communes (1937) **Agrarian Reform of 1964**
T1 (1972-1991)	First oil roads built, first exploitation of oil fields Apogee of both colonization and petroleum exploitation American Texaco and National Petroleum Corporation (formerly CEPE, now Petro-Ecuador) engaged as a consortium in oil activities.	Yasuní National Park (1979) Cuyabeno Biological Reserve (1979) Limoncocha wetland reserve (1985) Pañacocha Protected Forest (1986)	**Agrarian Reform of 1973** Ley of prevention and contamination control (1976) (it was not effectively implemented) **Forestry Law and conservation of protected areas (1981)** Law of colonization of Amazon (1978) Law of "indigenous nationalities" (1985)
T2 (1992-2001)	The state-owned Petro-Ecuador company assumed oil production and data compilation on gas produced was requisite. Oil companies were under constant but informal surveillance of non-governmental organizations, and environmental activist groups.	Sumaco-Napo Galeras National Park (1994)[22] Patrimony Forest (1993) Ministry of Environment is created Intangible Zones (ZITT) (1999)[23] Biosphere reserve declared by UNESCO (2000) Cofán –Bermejo Ecological Area (2002)	The Texaco trial (1993) Remedial Action Plan (RAP) **-** (1995) Environmental Management Act (1999) Environmental Regulations for Hydrocarbon Operations (RAOHE, 2001) Regulation for indigenous people consultation and participation (2002) Contamination Prevention and Control Act (2004)
T3 (2001-2012)	The state-owned Petro-Ecuador company assumed oil production and flared and vented associated petroleum gas reporting to Ministry of Environment (MAE) is mandatory.	Geographical definition of the ZITT (2007) Yasuní –ITT Initiative (2007) Program Socio-Bosque (2008) San Carlos Protected Forest (2011) Payamino Protected Forest (2011)	**Constitution of 2008** **Declaration of Nature Rights 2008.** **Art. 407**: extractive activity of non-renewable resources, including logging, is prohibited in protected areas and areas declared unbreakable[24].

[22] The National Park is located mainly outside the study area, only small fraction is within (see chapter 3)

[23] The Intangible Zones (ZITT) were zones for the uncontacted peoples Tagaeri and Taromenane. ZITT was not geographically defined until 2007 (Pappalardo et al., 2013).

[24] These resources may occasionally be exploited pursuant to a well-defined request by the President of Ecuador and a prior declaration of national interest by the National Assembly, which, if it deems it necessary, may convene a popular consultation.

Table 5. **(continued)** The three main different periods recognized during the crude oil production in Ecuador, compared with main legal events and conservation measures. Table is adapted from Juteau et al., (2014) and Saqalli et al. (2018).

Time period	Oil activities and type of management	Conservation measures	Main legal actions and remediation plans
After 2012	Petro-Ecuador engages in majority of the oil production. Ecuador voluntarily engaged in zero flaring goals.	Yasuní –ITT Initiative fails (2013)	**Reform of the hydrocarbons law (2009)** **Art. 94:** 12% of surplus production allocated to GADs. Approbation 10/03/2013 of the exploitation of Yasuní-ITT block. Texaco loses trial in Ecuador (2013) **Inter-Ministerial Agreement No. 33 (2013):** Implementation of the Plan for the Protection of Isolated Indigenous Peoples. Texaco wins trial in The Hague (2018) Programme of Environmental and Social Remediation (MAE-PRAS, 2016)

The thesis focuses mainly on T3, being the most recent period succeeding the establishment of hydrocarbon regulations[25]; and documentation that indicate the retrieved data is of improved quality and therefore more reliable to use for risk assessments, according to the National Secretary of Planning and Development in 2013 (SENPLADES, 2013).

To evaluate the importance of three key hazardous sources, a three question online-survey was conducted, regarding the perceived overall impact weight on societal, economical, health, and environmental assets. The first question was: “what is the relative weight of oil spills, gas flaring and oil pits as potential contaminating sources in the total pollution impact?” The working sector background of the people interviewed was the second question, formulated as: “In which sector do you work?”

[25] RAHOE for its abbreviation in Spanish

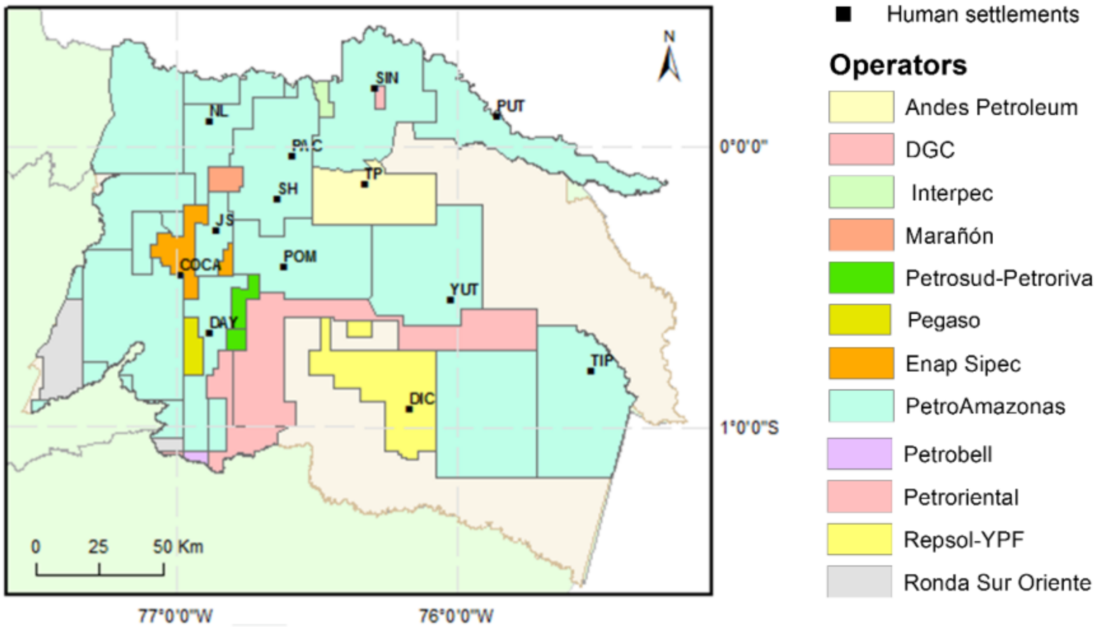


Figure 7. Oil blocks in the North-eastern Ecuadorian Amazon. Programme of Socio-environmental Remediation of the Ministry of Environment of Ecuador (MAE-PRAS, 2018).

All weights were measured on a scale from 1 (lowest) to 5 (highest). The resulting survey had 25 out of 100 respondents, corresponding to different activity sector stakeholders. The represented sectors were academia/research, non-governmental organizations, public governmental institutions and oil operators. The responses tended to give a similar weight to the different sources, i.e. oil pits were perceived as slightly more important (3.75), followed closely by oil spills (3.6) and gas flaring (3.4). The perception was that hazard sources weight do not vary significantly among them (Appendix A1) and there is lack of consensus between actors (Appendix A2), meaning that further in-depth surveys and other methods should be employed. In the meanwhile, the three sources should be investigated, as this survey suggests they are equally potential hazards to the NEA[26].

[26] See Appendix A for details on methodology and detailed results of this survey.

1.3. Thesis rationale and research questions

At least, two-thirds of the data used in this thesis were compiled from online public data, from the National Board of Hydrocarbons, and from the ANR-MONOIL and MAE-PRAS research collaboration agreement. Two assets were selected as case studies for environmental vulnerability assessment: (1) natural heritage and biodiversity, and (2) groundwater resources. This choice was made based on the relevance of these attributes of concern, in general (Al-Adamat et al., 2003; Gardner et al., 2009; Sala et al., 2000; Shrestha et al., 2017) and for the studied area (Barraza et al., 2018; Mena et al., 2006; Province GAD Orellana, 2011; Province GAD Sucumbios, 2013; SENPLADES, 2013; Wernersson, 2004). But this was also was guided by practical criteria: time and budget constraints, availability, accessibility and perceived completeness and quality of data.

The Ph.D. thesis is framed within the trans-disciplinary ANR-MONOIL project, in partnership with Ecuadorian institutions (MAE-PRAS, 2016; MONOIL, 2017) and the National Secretariat of Education, Science and Technology of Ecuador (SENESCYT). This thesis aims to answer three main questions:

1. What are the spatial and temporal variations of hazardous emissions at the sources of pollution? For this purpose, (i) oil spills, (ii) gas flaring discharges and (iii) deposited total petroleum hydrocarbons in unlined oil pits were estimated.

2. How can we map current environmental vulnerability levels based on (i) groundwater and (ii) natural heritage, including biodiversity, to address their spatial variations?

3. How can we implement ERA using quantitative estimates of oil hazards and environmental vulnerability, coupled with overlay-index methods? This was illustrated with the evaluated risk of unlined pits to groundwater.

In addition, an underlying problem is addressed throughout the chapters: How suitable are institutional public data for risk assessments in developing countries?

For this purpose, hazards from oil infrastructure first had to be evaluated. Oil spills, unlined oil pits and flaring equipment have been considered. The hazards were assessed through the estimation of emissions at the pollutant source, in other words, at each oil infrastructure that is potentially capable of accidental or intentional release of pollutants. The frequencies and amounts of these emissions had not been previously estimated across the NEA. They needed to be addressed, before any subsequent hazard or exposure evaluation could be intended. The thesis has been divided in four chapters:

Chapter I, entitled: "Spatial analysis of accidental oil spills using heterogeneous data: a case study from the North-Eastern Ecuadorian Amazon" is a ***publication in revision*** that intends to evaluate hazard from oil spills; several datasets were disaggregated and categorized by oil infrastructure prone to oil spills that can potentially contaminate soils and waters, notably accounting for the spatial location of these infrastructures within oil blocks which allowed to pinpoint the heterogeneous quality of publicly available data.

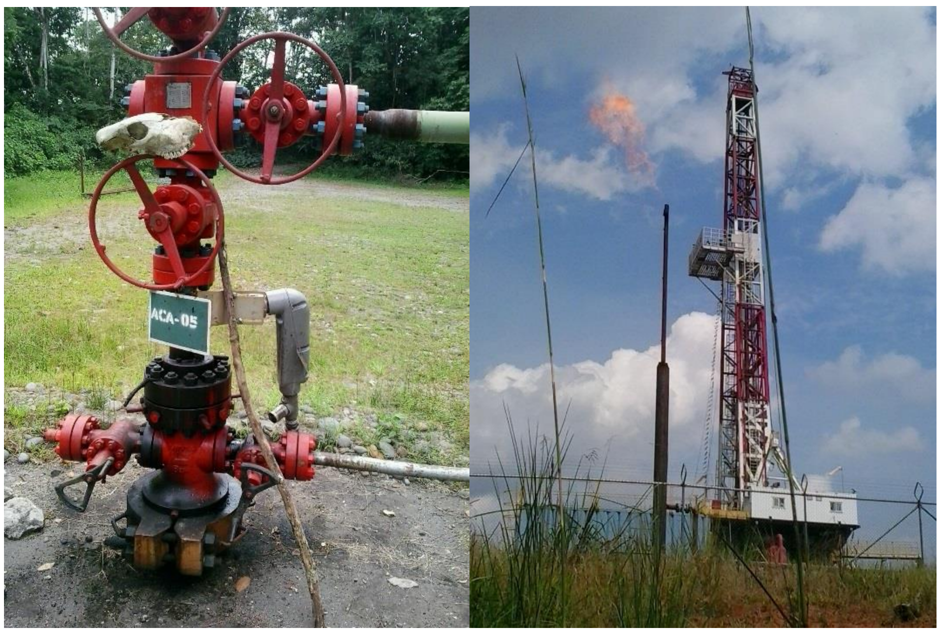
Chapter II, is entitled "Spatial inventory of selected atmospheric emissions from oil industry in Ecuadorian Amazon: insights from comparisons among satellite and institutional datasets", is a ***submitted publication*** that encompasses flaring infrastructure hazards. It is perceived as a high hazard due to the constant release of gases and particulate aerosols into the atmosphere.

Chapter III, is dedicated to environmental vulnerability, specifically the perceived important asset of biodiversity and natural heritage. The challenge of scoring, rating and spatially indexing the vulnerability of this asset is apprehended and discussed. Its title is "Spatial vulnerability assessment of natural heritage and biodiversity using current land use cover and nature protection levels".

Chapter IV is a ***publication in preparation*** that apprehends the last component of an ERA, the combination of indexed vulnerable assets and hazard from crude oil production to provide risk maps. There are many combinations that could be evaluated. Due to time limitation, the risk assessment was exemplified using only one environmental asset, but another important one: groundwater. It was scored, weighted and indexed spatially, and then combined with the unlined oil pit spatial distribution which logically appeared as the most likely to contaminate groundwater. The title of this final chapter is then **"**Risk assessment of unlined oil pits to groundwater in Ecuadorian Amazon: A modified GIS-DRASTIC approach".

Finally, the thesis presents a **general discussion**, **conclusion** and **perspectives** for future research.

Chapter I - Spatial analysis of accidental oil spills using heterogeneous data: a case study from the north-eastern Ecuadorian Amazon



Left image: an active oil well. Right image: a well being drilled (Auca oilfield)

Article published in *Sustainability*

Abstract

Accidental oil spills were assessed in the north-eastern Ecuadorian Amazon, a rich biodiversity and cultural heritage area. Institutional reports were used to estimate oil spill volumes over the period 2001-2011. However, we had to make with heterogeneous and incomplete data. After statistically discriminating well- and poorly-documented oil blocks, some spill factors were derived from the former to spatially allocate oil spills where fragmentary data were available. Spatial prediction accuracy was assessed using similarity metrics in a cross-validation approach. Results showed 464 spill events (42.2.yr^{-1}.), accounting for 10,000.2 t of crude oil, equivalent to annual discharges of 909.1 (± SD = 1,219.5) t. Total spill volumes increased by 54.8% when spill factors were used to perform allocation to poorly-documented blocks. Resulting maps displayed pollution "hotspots" in Dayuma and Joya de Los Sachas, with the highest inputs averaging 13.8 tons km^{-2} yr.-1. The accuracy of spatial prediction ranged from 32 to 97%, depending on the metric and the weight given to double-zeros. Simulated situations showed that estimation accuracy depends on variability in incident occurrences and in spill volumes per incident. Our method is suitable for mapping hazards and risks in sensitive ecosystems, particularly in areas where incomplete data hinder this process.

Keywords: spatial prediction; hydrocarbons; spill estimates; the Amazon; pollution hotspot

Highlights

- Heterogeneous oil spill data were used to map hazards in sensitive ecosystems.
- A spatial inventory of spills was completed using spatial allocation techniques.
- The reliability of the approach was assessed through cross-validation.
- Long-term spill contamination is an effective proxy method for mapping hazards.

I.1. Introduction

Several routine operations in upstream oil and gas production pose risks due to hazardous emissions including gas release and accidental oil spills (Jernelöv, 2010). The latter, which can be attributed to pipelines, oil wells/platforms and processing oil separation batteries, result from both regular operations and severe incidents (Burgherr, 2007). Accidental oil spills can have an impact on soil and water, and consequently on the health of local populations and agricultural production (Chang et al., 2014). Toxic chemical contaminants, such as Polycyclic Aromatic Hydrocarbons (PAH) and heavy metals contained in oil, can leach into the soil, reach drinking and groundwater, and damage vulnerable ecosystems (Bondur, 2011; Wernersson, 2004).

The north-eastern Ecuadorian Amazon (NEA) is considered as a biodiversity hotspot, with sensitive ecosystems, comprising a large number of endemic species of flora and fauna (Bass et al., 2010; Myers et al., 2000). Rich oil reserves in the region boosted economic growth, and helped improve education and health services (Larrea, 2006). But local communities in the NEA claim the benefits for them were negligible, while they endured decades of oil-related pollution, which caused serious health conditions and environmental degradation (Baynard et al., 2013; Butt et al., 2013; Finer et al., 2008). Significant adverse effects include: declining livestock health (Waldner et al., 2001); severe health problems in local communities and society (Chang et al., 2014; San Sebastián et al., 2001; San Sebastián and Hurtig, 2005); degraded environmental amenities; a drop in the value of rural land (Boxall et al., 2005); and biodiversity loss (Finer et al., 2008; Jernelöv, 2010). These claims resulted in an international trial between local communities and the international oil firm Texaco-Chevron which is still ongoing (Buccina et al., 2013; San Sebastián and Hurtig, 2005). The NEA is consequently a relevant case study for addressing region-wide oil contamination, and it also raises the issue of how to assess spill contamination from a scientific point of view, as these are mainly represented by small (≤700 t) crude oil discharges distributed in space and time (Burgherr, 2007).

Ecuador’s program for social and environmental reconstruction (PRAS) put major effort into compiling a historical oil spill geo-database for remediation purposes. Other entities, including the Ministry of Energy and Mines (MEM) and the Amazon Defense Front (ADF, a non-governmental organization) have also attempted to estimate the spills that occurred in the NEA from 1972 to 1991. Three distinct periods of management in Ecuador have been identified (Juteau et al., 2014):

- T1 (1972-1991): the foreign company Texaco and the National Petroleum Corporation (CEPE) were engaged in oil & gas activities. Over this period, the MEM estimated that 397,5 x 106 t of crude oil were discharged to the environment (Amazon Defense Front, 2008).
- T2 (1992-2001): State-owned Petro-Ecuador took over oil production. Data compilation on accidental oil spills was reinforced.
- T3 (from 2001): period after the environmental decree taken for regulating oil activity (RAHOE).

Spatial gridding of disaggregated oil spills is useful for several purposes, including: pollution hotspot analysis, trajectory modelling, mapping of hazards and exposure, multiple-pollutant and risk assessments (Andreo et al., 2006; Guttikunda and Calori, 2013; Lahr et al., 2010; Lahr and Kooistra, 2010). Only a few studies have attempted to make spatially explicit predictions of oil spills using kernel density estimation methods, with corrections based on data from surveillance efforts (Serra-Sogas et al., 2008) and including human and other external factors associated with oil spill patterns (Bertazzon et al., 2014). Oil spill studies are typically performed in an offshore context, and only very few have looked at onshore events (Burgherr, 2007). And none of them included reliability analyses or validated models for the predicted patterns. Because institutional databases tend to be incomplete and heterogeneous, no study has ever attempted to use them to spatially estimate oil spill volumes in order to improve hazard or risk assessments.

The aim of the present study was to analyse oil spill patterns in the NEA. It focused on the T3 management period, in which monitoring was much improved compared to the previous periods (SENPLADES, 2013). Our aim was also to focus on recent contaminations to obtain up-to-date estimates of potential hazards. Surveys of local populations were used to determine spatial locations of oil spill sources and also how the contamination potential of oil spills is currently perceived (Maestripieri and Saqalli, 2016), but these results require a more in-depth quantitative analysis. The present study builds a regional oil spill inventory and draws up spatially explicit oil spill maps. In parallel, the issue of incomplete and heterogeneous databases is addressed, so as to obtain estimated spill values in areas where data are incomplete. Finally, the reliability of our estimation procedure was analysed using a cross-validation method, and the factors most likely to influence the quality of our estimates were assessed using simulated data.

I.2. Materials and Methods

I.2.1. Study area

The study was restricted to the provinces of Sucumbíos and Orellana in the NEA (~144-900 m.a.s.l., Amazon lowlands), representing a surface area of 35,051 km^2 (Figure 8A). Upstream oil & gas production infrastructures are found in this zone, and associated potentially polluting activities take place there (Figure 8B). The North-eastern Ecuadorian Amazon (NEA) is a sensitive Amazon forest, a biodiversity hotspot. For an illustration purpose, at least 210 mammals, 131 amphibians, 558 birds and 3,213 vascular plant total species richness, have been reported only in Yasuni National Park (Bass et al., 2010; Finer et al., 2008).

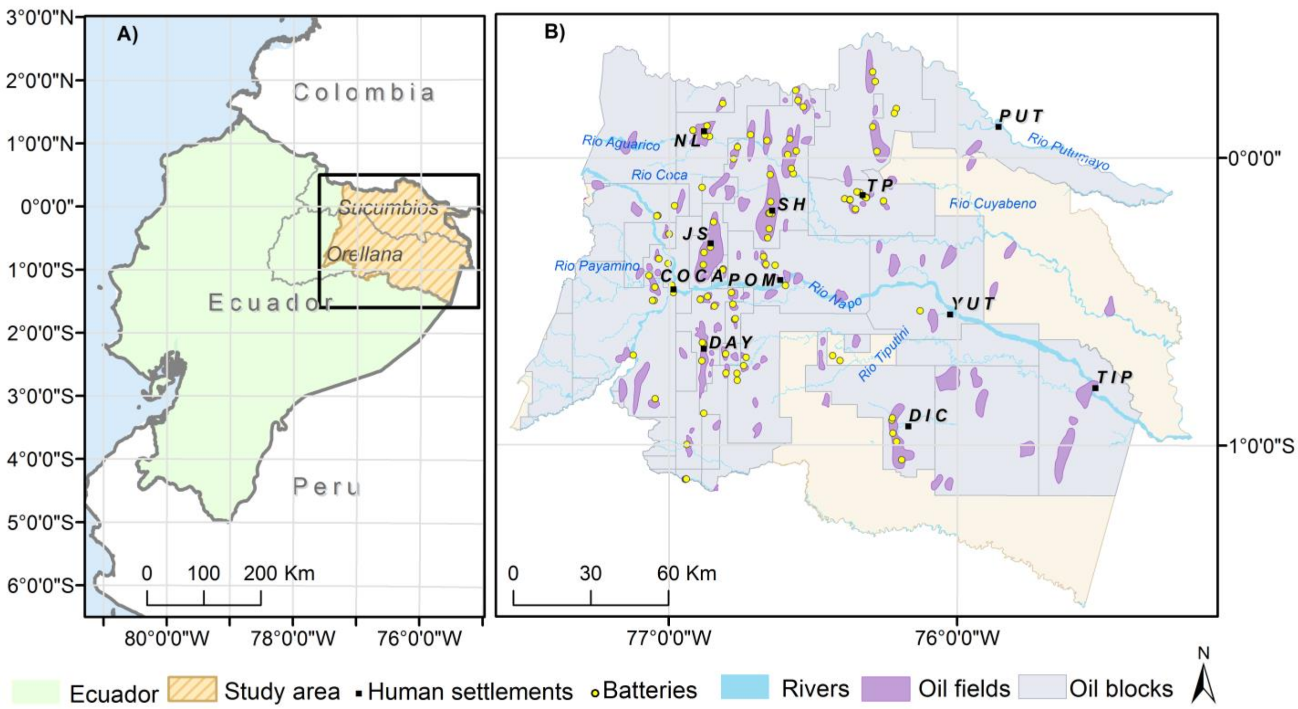


Figure 8. (A) The NEA in Ecuador; (B) study area within the two NEA provinces of Sucumbíos and Orellana including oil fields and oil separation batteries. The names of the human settlements represented are abbreviated: DAY= Dayuma; NL = Nueva Loja (aka. Lago Agrio); SH = Shushufindi; TP = Tarapoa; PUT = Putumayo; COCA = Puerto Francisco de Orellana; JS = Joya de Los Sachas; YUT=Yuturi; DI=Dícaro; TP=Tiputini.

I.2.2. Data for crude oil spills

In this study, oil pollution was quantitatively estimated and spatially allocated using GIS and institutional inventory databases, including the location of oil infrastructures and oil spills. Drilling platforms are positioned directly over producing oilfields, whereas oil separation batteries and refineries are located at Nueva Loja and Shushufindi (Figure 8). Distribution pipelines connect the oil wells to oil separation batteries, which are cylindrical or spherical vessels used to separate oil, gas and water from the total fluid stream produced by a well. Oil separation batteries are in turn joined to the main pipeline network so that products can be transported to oil separation batteries before being sent west, across the Andes Mountains to the coast for export. These data were obtained from PRAS or public online databases (Table 6). The data were disaggregated and categorized annually for the T3 period (more specifically for the 2001-2011 period, for which data were available).

Table 6. Databases used for spatial estimation of oil spills.

GIS data	Description	Sources
Oil blocks, Oilfields	Polygons, 1:250,000	National Board of Hydrocarbons (2014) Petro Ecuador (2013)
Oil wells, Platforms, Refineries, Oil separation batteries	Point	PRAS (1972-2014)
Oil pills	Point	PRAS (1972-2014)
Pipelines	Line	SENPLADES (2009)
Inventories		
Oil spills	Historical data	PRAS (2014) and local governments

I.2.3. Accounting for heterogeneity in data quality: well- vs. poorly-documented oil blocks

In Ecuador, the central government organizes bidding rounds to award oil blocks to operators, who then each implement different management plans. This results in non-uniform environmental disclosures, meaning that oil spill reporting can vary greatly between oil blocks.

There is indeed a strong contrast between some blocks with a high number of incidents recorded and others, sometimes comprising a large number of infrastructures, where few or no incidents have been reported.

It was therefore necessary to distinguish between well-documented and poorly-documented blocks. Assuming that incidents on a single infrastructure occur at a constant rate and are independent of one another, the number of incidents over a given time period can be described by the Poisson distribution's Equation 2. The probability of n incidents occurring during the study period is therefore:

$$P(n) = \frac{\lambda^{n}\, e^{-\lambda}}{n!} \qquad \text{(Eq.2)}$$

Where:

n = number of oil spills on the block from 2001 to 2011.

λ = expected number of oil spills from 2001 to 2011 calculated based on the number of oil infrastructures on the block and the oil spill rate for all the oil blocks in the study area.

Parameter λ was estimated using oil well data; wells account for 70% of oil infrastructures and are therefore considered as representative of all the infrastructures in the area. Oil blocks were categorized as poorly-documented (i.e. the probability of observing such a low number of oil spills over the time period was highly unlikely) if $P(n \leq nobs.) < 0.05$, and as well-documented otherwise.

I.2.4. Calculating the oil spill rates to be used for estimations on poorly-documented blocks

The PRAS datasets actually compiled certain attributes such as accident cause, age of oil well first drilled and private/state operator data, that is useful to investigate the cause of oil spills thus for future prediction of oil spills. The oil spill rates for poorly-documented blocks were considered inaccurate and were therefore replaced by the rate found as described above, using data from well-documented blocks. Based on the raw data obtained from these blocks, the annual average spill volumes were calculated separately for three types of infrastructures: oil wells, oil batteries and pipelines. These rates were expressed per infrastructure unit for oil wells and oil batteries and per kilometer for pipelines, and were subsequently used to estimate spills within poorly-documented blocks, assuming a constant incident risk across the study area.

The possibility of using spatial information to improve the accuracy of this oil spill rate depending on infrastructure location (e.g. proximity to human settlements might entail greater surveillance efforts) was examined through a geo-statistical approach. For this purpose, the Getis-Ord (Gi) statistic and Moran's I index value were calculated. The Gi assesses the degree of clustering (Getis and Ord, 1992) and the Moran's I index value made it possible to assess the spatial dependence of values, or autocorrelation. This index ranges from +1 to -1: positive values indicate clustering, negative values indicate dispersion and values close to 0, complete spatial randomness (Meng, 2015).

I.2.5. Oil spill mapping

In order to represent crude oil spills in a GIS environment, a grid with cells of 5x5 km was chosen to plot the spills and incorporate line and point infrastructure sources in the region. Spatial data processing was performed in ArcGIS®.

Two maps were created, one plotting actual oil spills from recorded events, a second taking into account heterogeneity in data quality and plotting estimated spills based on rates from well-documented blocks. This second map is a plausible spatial representation of oil spills.

Estimated spills were thus allocated to all infrastructures within the study area (excluding infrastructures that were not in production from 2001 to 2011). Estimated oil spill volumes at single point and line sources were added together within each grid cell to obtain total oil spill volumes per square kilometer and per year (Andreo et al., 2006; Burgherr, 2007). Values were mapped according to geometric intervals.

I.2.6. Validity of the procedure to estimate oil spills on poorly-documented blocks

The reliability of the method used to estimate oil spills on poorly-documented blocks was assessed using a cross-validation procedure, drawing on data from well-documented blocks. This dataset was divided into two parts: the first would be used to compute spill factors for the various infrastructure types (i.e. the training subset), the second to estimate the oil spill volumes within each cell and to then compare these estimated values with those actually observed (i.e. the testing subset).

Similarity indices were used to compare estimated vs. observed values. For this purpose, two different metrics were used, both designed for quantitative data and ranging from 0 (extremely dissimilar samples) to 1 (identical samples). The first one, Gower's coefficient (Gower, 1971), is symmetrical as it considers double zeros as a similarity (i.e. a "negative match"). The second one, the Steinhaus coefficient (Legendre and Legendre, 1998), is asymmetrical as it is not influenced by double zeros, and therefore gives greater emphasis to cells where oil infrastructures are located. In addition, the Pearson correlation coefficient was also calculated.

This procedure was repeated 1,000 times, using a random allocation of cells either to training or test subsets. It was therefore possible to calculate average similarity coefficients and their standard deviations. To determine the significance of these similarity coefficients, a Monte-Carlo procedure was performed, in which similarity coefficients were also calculated after observed and estimated spill volumes were randomly matched in order to generate a null distribution of the similarity/correlation coefficients (10,000 permutations computed).

Finally, to analyze to what extent the similarity coefficients were influenced by the variability in oil spill events, a simplified simulation-based approach was implemented, focused on oil wells. Using a uniform distribution to allocate oil spills to the wells, and a log-normal distribution to simulate the volumes spilled (which was the best choice with regard to the distribution of the real oil spills observed, whose volumes spanned four orders of magnitude (Limpert et al., 2001)), various datasets were randomly generated including different percentages of wells where no events had been reported over the given time period and different standard deviations of volumes spilled in each event. The σ parameter of the log-normal distribution for spill volumes was set between 0.0 and 3.0 in the various simulations. The μ parameter was set at 3.5 in simulations where spills had occurred on all the wells, and this value was changed accordingly in subsequent simulations to take into account the increase in the percentage of oil wells where there had been no spills.

I.3. Results

I.3.1. Oil spills: temporal and spatial patterns

Historical data indicate that oil spills occurred before 2001 and represented a total spill volume of 20,386 t. From 2001 to 2011, 464 accidental oil spills accounting for a total amount of 10,000.2 t were identified and documented by the PRAS of the Ministry of the Environment of Ecuador. 41.3% of the total volume of oil spilled was reported to have occurred in the T3 period, the focus of our study. The number of oil spills per year decreased from 2007 to 2011 for all infrastructures. Figure 9 presents the overall annual number of incidents and spill volumes.

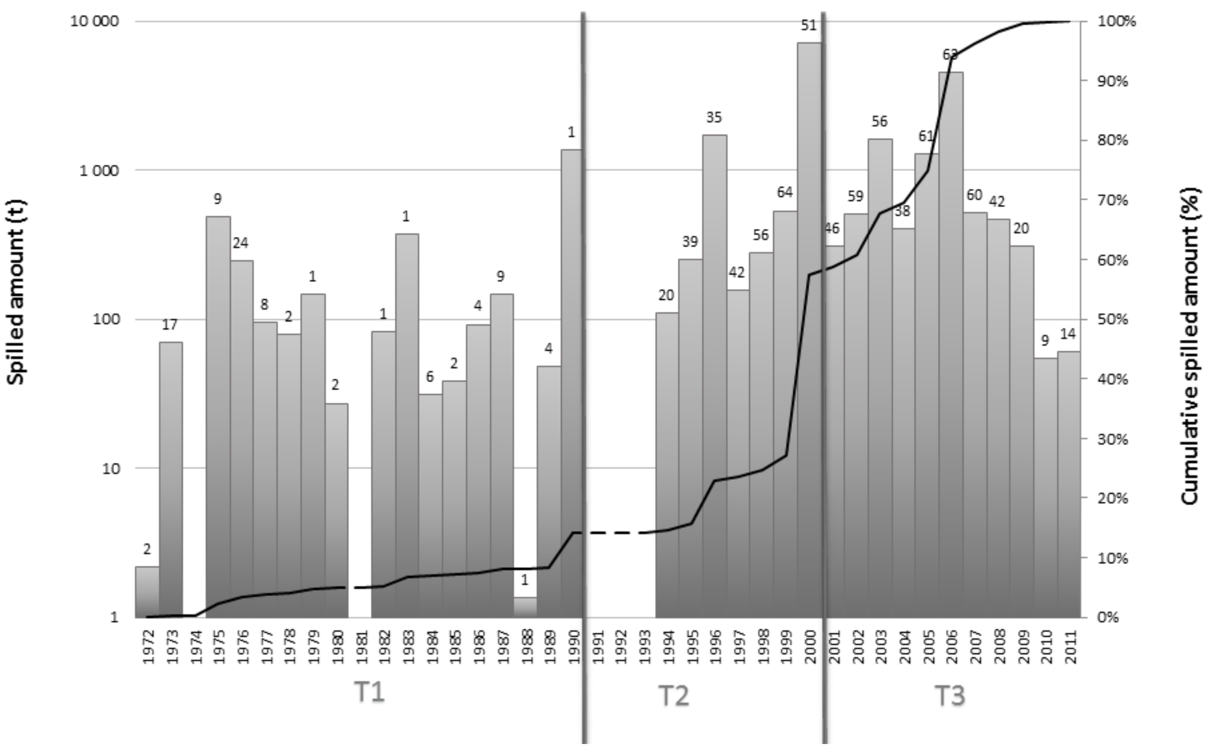


Figure 9. Historical number and volumes of oil spills. Number of accidental oil spills per year (individual values shown at the top of each bar) and the corresponding cumulative percentages relative to the total spill volume over the whole period. No datum was recorded from 1991 to 1993 (departure of Texaco).

Table 7 summarizes the spatial and average values in the number of accidental oil spill and spill volumes from 2001 to 2011. The different blocks give a total spill volume of 10,000.2 t, associated either with oil wells/platforms (6,971.7 t, n = 339), oil separation batteries (2,555.7 t, n = 107) or pipeline infrastructures (472.9 t, n= 7). Oil wells/platforms therefore contribute to ca.70% of oil spill volumes, while the 605 km of pipelines recorded account for less than 5%. Four oil blocks account for > 90% of the spill volumes, namely Auca, Sacha, Libertador and Lago Agrio.

Table 7. Total and average values on original oil spill data. A total of 13 oil blocks are located within the study area, and considering all the infrastructures only 4 of the 13 oil blocks present greater volumes. The subtotals are the results from a single oil block. Oil blocks by abbreviation and geographical location are shown as follows: AU = Auca, LA = Lago Agrio which are located in Pacayacu parish, LT = Libertador in Shushufindi, SC=Sacha in Joya de Los Sachas.

		Oil spill occurrences		Oil spill volumes		
		Total (n)	Average ($n.yr^{-1}$)	Total (t)	Average ($t.yr^{-1}$)	Share (%)
on oil blocks (N = 13)	block 61 (AU)	76	7±5.82	4,054.50	368.59 (±1,119.09)	40.54
	block 60 (SC)	98	9±7.8	2,801.56	254.68 (±324.87)	28.01
	block 57 (LT)	154	14±7.5	1,792.13	162.92 (±159.80)	17.92
	block 56 (LA)	53	5±3.6	368.78	33.52 (±47.23)	3.69
	sub-total	381	34.6±7.8	9,016.97	819.73 (±1,170.14)	90.17
on single infrastructure	Wells	339	30.82±0.19	6,971.66	633.79 (±1,149)	69.71
	Pipeline	7	0.6	472.88	42.98 (±55.4)	4.73
	Oil separation batteries	118	10.7±0.62	2,555.7	232.33 (±327.69)	25.56
	Total	464	42.2	10,000.23	909.11 (±1,219.46)	100.00

Figure 10A shows the locations of point sources, i.e. oil wells (n = 668) and oil and gas batteries (n=108), and line sources: 1,596.37 km of recorded pipeline. Most of the infrastructures are located in the north-west of the country, in the cities of Nueva Loja, Joya de Los Sachas and Shushufindi.

No spatial patterns could be identified with the metrics used (neither at block nor regional scales), neither in terms of spill volumes (Gi = 0.0028, Z = -0.66, P = 0.51; I Moran’s index =

0.006, Z = 0.56, P = 0.57) nor incident occurrences (Gi = 0.0038, Z = - 0.14, P = 0.89; I Moran's index = -0.024, Z = -0.84, P = 0.39), and no infrastructure attributes (except for typology i.e. well, battery or pipeline) could be significantly correlated with oil spill rates. The best option for estimating oil spill volumes on poorly-documented oil blocks was therefore to use average constant allocation drawing on spill rates from properly-documented blocks.

The following two maps: (1) spills actually recorded (Figure 10B) and (2) spatially-allocated estimates (for poorly-documented oil blocks) associated with recorded spills (for well-documented blocks) (Figure 10C) were used to investigate oil spill patterns. These maps show different levels of spills, mainly concentrated in the four oil blocks already mentioned (Table 7 and Figure. 10B), and highlight spill hotspots (Figure 10B). Spatial allocation of oil spills to poorly-documented oil blocks naturally resulted in an increase in total spill volumes.

Our final spatial distribution map shows that a hypothetical total of 15,481.5 t of oil was spilled in the NEA i.e. a 54.8% increase compared to the recorded total (Table 7). The variations in spills on the harmonized map for each type of infrastructure are: 4,494 t for oil wells/platforms (82.0% of the total increase), 835 t for oil separation batteries (15.2%), and 153 t for pipelines (2.8%). The cities of Tarapoa, Yuturi, Dícaro and Puerto Francisco de Orellana in the east all displayed larger oil spill volumes than in the original dataset (Figure 10C).

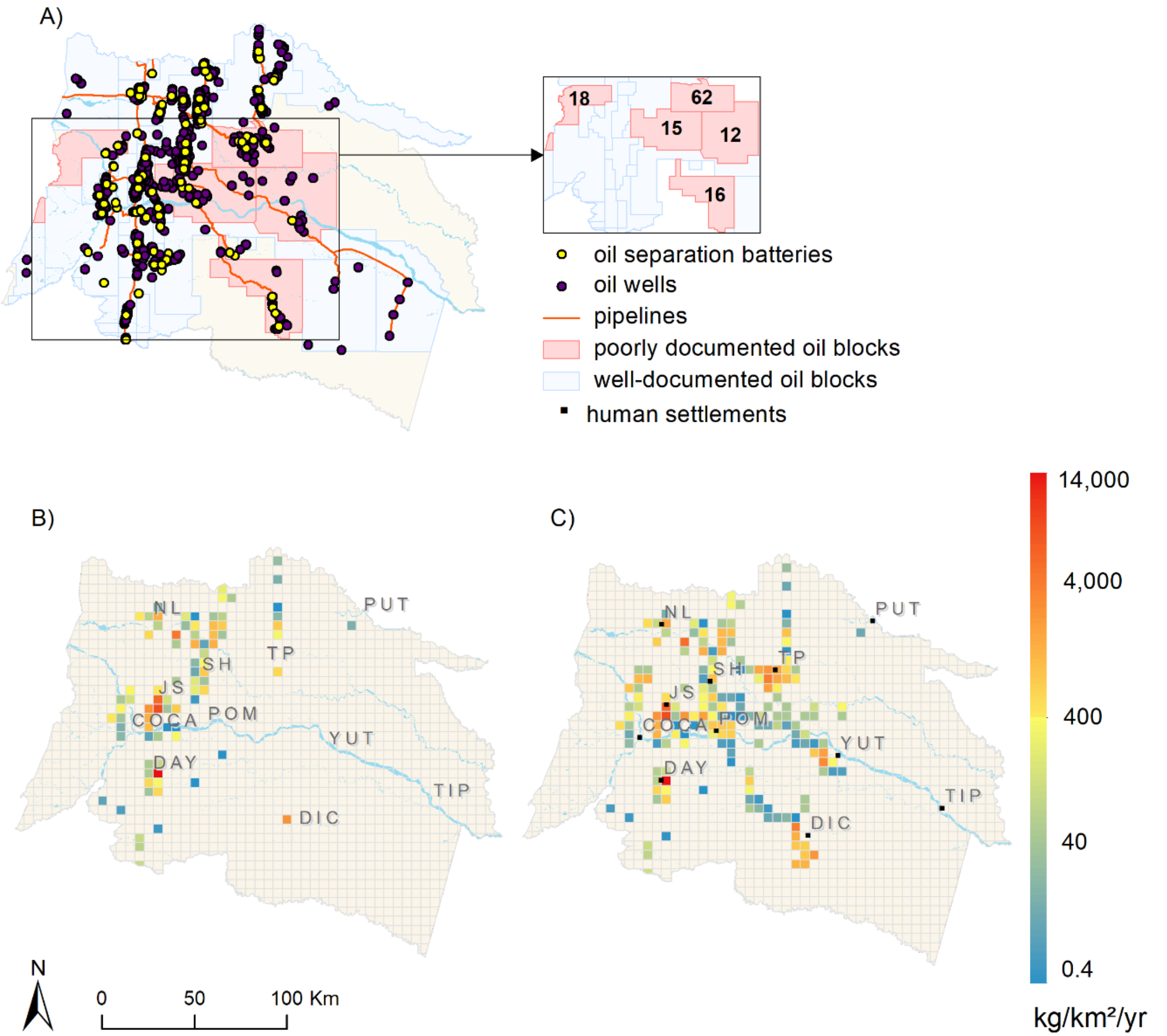


Figure 10. Spatial variations in oil spills during the T3 management period (2001-2011). (A) Map of the NEA showing the main oil infrastructures with well-documented (blue) and poorly-documented (red) oil blocks. (B) Originally recorded spills. (C) Harmonized map plotting data from (B) for well-documented oil blocks and average constant allocation to poorly-documented oil blocks.

I.3.2. Reliability of the procedure used to estimate missing data

Although assessing the validity of our approach on presumably poorly-documented oil blocks was impossible due to the age of the oil spills, difficulties in accessing the fields and the absence of any further data, it was however feasible to evaluate the reliability of the method implemented using data from properly-documented blocks. Similarity metrics between estimated and observed spill volumes across the 942 grid-cells from well-documented blocks amounted to 0.97, 0.58 and 0.32 for Gower's coefficient, the Pearson correlation coefficient and the Steinhaus index respectively.

The high average value obtained with Gower's coefficient was related to the large number of null values in the dataset and to the symmetrical nature of this index, giving some weight to double-zeros. Such a high Gower index value is actually very likely to be obtained only by chance, as shown by the Monte-Carlo procedure (10,000 permutations, $P = 0.84$). In contrast, the Steinhaus index does not take double-zeros into account in similarity computations, and therefore focuses on the grid-cells where oil infrastructures are located. Although not very high, its value is significant according to the results of the Monte-Carlo procedure (10,000 permutations, $P = 0.0016$), as is the value obtained with the Pearson correlation coefficient (10,000 permutations, $P = 0.0087$).

By using an exploratory approach on simulated data, we were able to investigate to what extent our predictive ability would be affected by the sources of spatiotemporal variability in oil spill patterns. Figure 4 shows that the similarity between observed and estimated spills obtained with the Steinhaus index is high (i.e. above 0.8) when both incident occurrence and spill volumes are highly predictable. In contrast, even if spill volumes remain predictable, this similarity drops when the stochasticity of events increases (Figure 11).

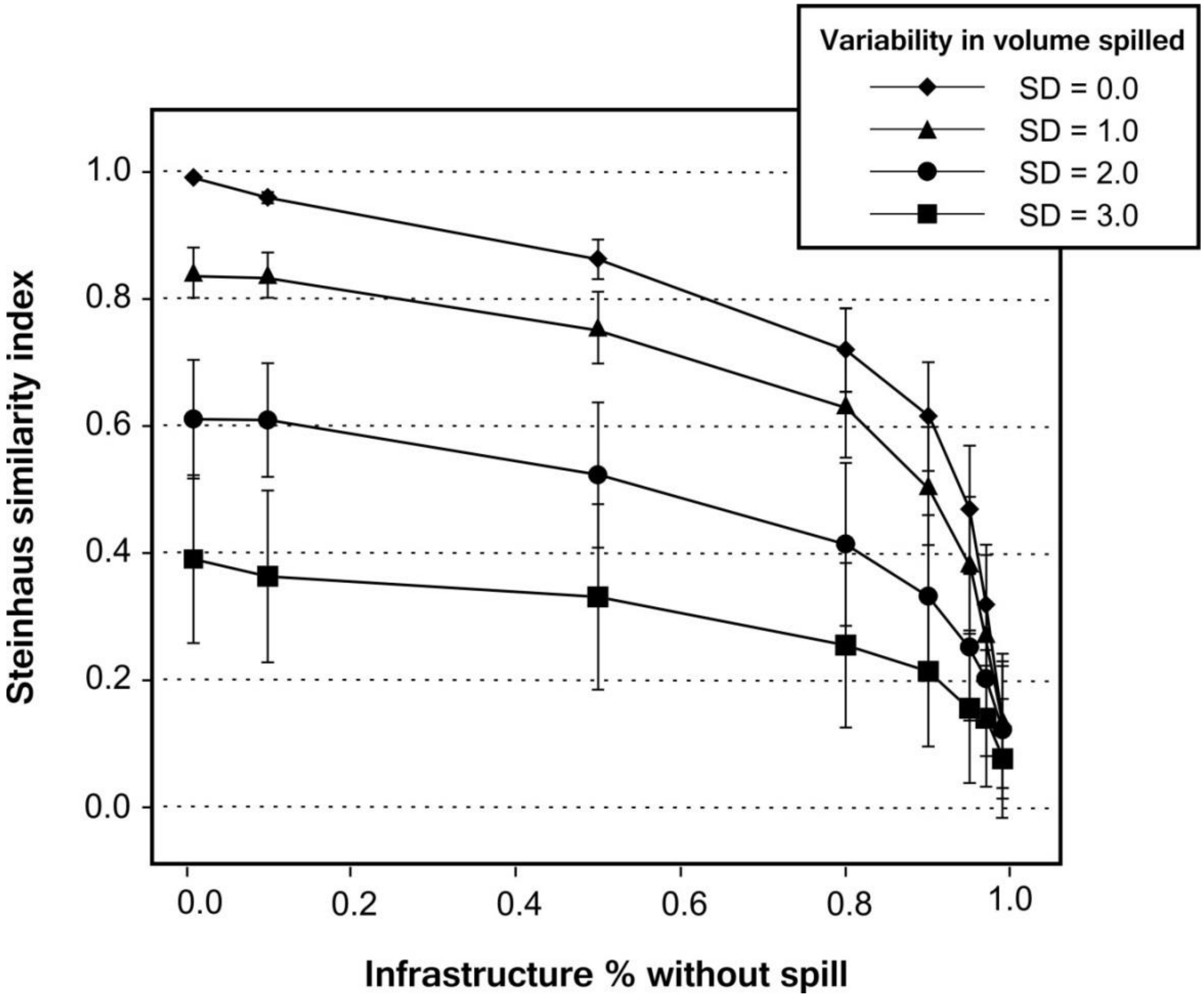


Figure 11. Variations of the Steinhaus index assessing the similarity between observed and estimated oil spill volumes across a simulated area of 23,550 km^2, as a function of different sources of heterogeneity. This highlights the influence of variability in incident rates (i.e. spills evenly distributed across infrastructures vs. concentrated among a small proportion of infrastructures) and in the volume of oil spilled per incident.

I.4. Discussion

I.4.1. Uncertainties and data quality

I.4.1.1. Data reporting

In the NEA, oil spills decreased in later years, in line with global trends (Schmidt Etkin, 2001). In keeping with other studies (Bertazzon et al., 2014; Burgherr, 2007; Serra-Sogas et al., 2008), this work suggests there are primarily two reasons for this: (1) activities are better managed and there is a general tendency for spills to decrease; and (2) surveillance efforts might be less consistent than in previous years because of reduced data disclosure. The maps provided in the present study are the most accurate representations of the spatial distribution of oil spills according to available data. In this respect, the PRAS dataset is a thorough compilation of environmental assessments performed by several local government agencies. The multiplicity of data sources is not an obstacle to running a comprehensive analysis of oil spill patterns, provided certain quality requirements are fulfilled (Burgherr, 2007). In the present study, the uncertainties related to using oil spills at specific infrastructure points are acknowledged and require caution when using these data. However, in a 5 x 5-km^2 grid, the effects of location errors are presumably less significant.

I.4.1.2. Spill estimates

Estimating the quantity of hydrocarbons accidentally discharged to the environment remains a difficult task, and requires good incident tracking/reporting. Data records are often not exhaustive because industry operators do not necessarily disclose information, and data disclosure can vary depending on the operator and their respective management plans (Buccina et al., 2013; Schweitzer, 2008). In this respect, it should be noted that most of the oil spills recorded occurred on blocks managed by the national state company. “Poorly-documented” oil blocks are categorized as such based on the assumption that data are incomplete, following statistical analysis. But the statistically significant lower occurrence of events on these blocks could also be related to more efficient management by oil operators. These different scenarios could generate alternative results, and it could be assumed that the proper distribution of oil spills lies somewhere between official and computed patterns (Figure 10B and 10C).

I.4.2. Accuracy of oil spill estimations

Incidents can occur because of infrastructure corrosion, human errors during oil production and transport, politically-motivated attacks, natural disasters, and so on (Burgherr et al., 2012; Darbra et al., 2010). The causes of oil spills have been documented in the PRAS database, and could have helped to improve the oil spill predictions. For instance, one hypothesis is that greater efforts are put into surveillance in the vicinity of settlements, leading to improved incident reporting, or that incident rates are higher in some areas than in others (zones subject to flooding or seismic activity). But this was not supported by the data and, more generally, exploratory spatial analyses of oil spills were not conclusive as no clustering or autocorrelation was found in spill rates or volumes. The only information that proved useful for refining predictions was the type of infrastructure (i.e. wells, batteries, pipelines), which enabled us to estimate different spill rates. As a result, our ability to predict oil spills within the study area is relatively limited, as shown by the cross-validation performed on presumably properly-documented oil blocks: when no weight was given to double-zeros (often corresponding to grid cells devoid of oil infrastructures), ca. only 30% of spatial variations in oil spill volumes could be predicted.

This value was much improved (reaching 97%) with Gower's coefficient, giving some weight to the cells where there is no oil infrastructure. This highlights the importance of choosing the right metric to assess similarity. Although the Steinhaus index is better suited in the present case to assess the quality of our estimation method, this may be relevant to use Gower's coefficient for assessing the reliability of the contamination patterns predicted at the regional scale, without excluding the trivial situations corresponding to the areas where there is no oil infrastructure and thus no oil spill.

The analysis of simulated situations has shown that the ability to predict oil spills is strongly dependent on unexplained variabilities in incident rates and spill volumes. In other words, a lower variability in spill patterns logically improves predictions. Low variability can be observed in other contexts, for instance when spills are mostly due to continuous infrastructure leaks (McKain et al., 2015). This also implies that any data-mining method allowing to reduce the part of residual variability in oil spill patterns will improve predictions. Beyond the methods used in the present study, other approaches such as machine-learning algorithms could potentially help to improve the estimation accuracy of spill factors, provided features of oil & gas infrastructures and/or spatial data are available and offer sufficient information about event probability or severity (Arellano et al., 2015)

I.4.3. Spatial distribution of spills and hazard potential

Pollution hotspots could be identified from the two oil spill maps (Fig. 10B and 10C). Joya de Los Sachas displays the highest level of pollution, but other sites also exhibit high levels of spills such as Yuturi, Dayuma, and the area from Shushufindi to Nueva Loja. Owing to the spill volumes allocated to poorly-documented blocks, Dícaro and Tarapoa appear as potential areas of concern too. Ongoing research aims to combine maps depicting the impacts to the different environmental compartments (atmosphere, pedosphere and hydrosphere) and classify them by pollutants (e.g. black carbon, total hydrocarbons, heavy metals) from different sources (accidental oil spills, mud drilling pits and gas flares).

Achieving the target of zero harmful discharges is a real challenge. However, our data analysis does differentiate sites where oil spills exceed hazard severity thresholds and where the potential economic, health and environmental losses are the highest. Severe oil spills are better defined when one or more of the following criteria are met, including the amount spilled, remediation costs, the impacted area and the environmental damage sustained in an single event (Burgherr and Hirschberg, 2014). Data in this study offered the opportunity to directly investigate the first criterion. For instance, an individual spill of 10,000 t is generally considered as a severity threshold; however, other standard international thresholds give lower amounts (34 -136 t) (Burgherr, 2007) comparable to spill volumes in the present study (Table 8). These hazard thresholds are exceeded in several locations across the NEA, but are not highlighted by our estimations and spatial maps, due to the grid resolution and time scale used. Other studies, however, suggest chronic pollution from oil spills in the NEA (Bissardon et al., 2013; Serra-Sogas et al., 2008), and the mapping of total pollution over a longer period (2001-2011) might better estimate hazard-prone areas in the case of low-volume but recurrent spill events.

Table 8. Number of emission events exceeding international severity thresholds over the period 2003-2012 in human settlements. The international standards selected are: the Environmental Technology Centre (ETC) and the Environmental Research Consulting (ERC). YUT, POM and TP are in poorly-documented blocks, meaning the number of events might be higher.

Location	abbr.	ETC (136 t)	ERC (34 t)

COCA	COCA	-	2
Dayuma	DAY	1	3
Dícaro	DIC	1	1
Joya de los Sachas	JS	8	21
Nueva Loja	NL	2	10
Pompeya	POM	-	-
Putumayo	PUT	-	1
Shushufindi	SH	-	6
Tarapoa	TP	-	1
Yasuní-ITT	TIP	-	-
Yuturi	YUT	-	-
Number		**12**	**45**

Oil spills by infrastructure type are in concordance with worldwide historical reports (Burgherr, 2007; Schmidt Etkin, 2001). The most severe spills originated from tankers during offshore activities. But this source aside, 8.7% of the spills recorded worldwide were due to pipelines and 29.3 % to oil wells/platforms. Similarly, data recorded in the NEA indicated 4.7% and 95.3% of spills, respectively. This goes against the common belief that oil spills are essentially due to pipelines (Jernelöv, 2010).

I.4.4. Potential economic, health, and environmental losses

A study performed by the International Oil Pollution Compensation Fund (IOPCF), founded on a historical oil spill dataset, derived a conversion factor of USD 51,437 per ton of oil spilled, based on costs associated with environmental damage, clean-up, and losses in the fishery, tourism and farming sectors (Kontovas et al., 2010). When this factor is applied to our case study, a sum of USD 514.4 million is obtained for the whole period considered (2001-2011), i.e. USD 46.8 million.yr^{-1}. (USD 72.4 million.yr^{-1}. if the predicted spill volumes in poorly-documented blocks are taken into account). This is an early reference cost as until now the factor was generally applied to datasets from offshore oil spills; further site-specific analyses need to be conducted in this sensitive area.

In the NEA, studies have revealed the acute toxicity of drinking water and sediment, increasing the vulnerability of freshwater ecosystems (Castello et al., 2013) and causing human health issues, ranging from dizziness to cancer (Burgherr et al., 2012; Burgherr and Hirschberg, 2014). Future studies should deepen understanding of other hazard sources (i.e. waste pits) at local and regional scales. Hazard maps could improve decision-making, when combined with vulnerability maps (e.g. groundwater, biodiversity values, etc.). Previous studies have assessed the risk of oil concessions merely by overlaying hazards with maps of vulnerable assets at larger scales, with a lower spatial resolution, and greater surface areas than the ones in this study (Baynard et al., 2013; Cuba et al., 2014; Finer et al., 2008). This study is therefore more comprehensive, and offers the potential for integrating spatio-temporal patterns of contamination; this represents a first step towards spill maps that could be used as input to model contaminant dispersion and trajectory, or to improve risk assessments at finer scales and spatial resolutions when associated with vulnerability maps.

I.5. Conclusion

This study presents a “bottom-up approach” to processing and visually representing datasets specific to upstream oil & gas production activities (i.e. type of infrastructure) in gridded form. In addition, the estimation of spills and the subsequent harmonized spatial distribution of pollutants in the NEA are of great importance for this sensitive area or other, as they potentially provide information about the hazards posed by these activities. Most importantly to address for such small oil spills reported with lack of data usually relevant for oil spills prediction. This forms a set of regional and local multiple-source spill data combining the latest available local information with allocation in space and time. Overall, the sum of incidents having occurred at oil separation batteries and oil wells/platforms during the T3 management period contradicts the common belief that they represent only a small share of spill sources compared to pipeline networks. The willingness of institutions and operators to disclose data is the key. Data disclosure could help improve evaluation of pollutant releases for more effective decision-making in land use planning to improve health, economic and environmental conditions. Finally, spill maps represent the total potential oil spill hazard, which could be useful for future risk assessments in context with scarce and reliable data to predict oil spills.

From the assessment of oil spills in well- vs. poorly-reported oil blocks to airborne emissions and their quality

In chapter I, several methods were used to homogenize oil spills to enable long term regional and homogenous emission mapping that could input to a more sophisticated model for hazard and risk assessment. Multiple infrastructures potentially discharging contaminants to the environment, either accidentally or intentionally such as the aforementioned wells, oil separating batteries, platforms and pipelines, were considered for analysis regarding data quality, public disclosure, and quantities per infrastructure type. A potential approach to measure spatial prediction accuracy was proposed. Spatial predictions explained more than 30% of the variability in spills in the vicinity of oil infrastructures. Although this value is not very high, and may be increased in the future by the inclusion of other explicative variables and/or via the use of other numerical analyses, this is significantly more than a random spatial allocation would do.

These emissions could potentially impact soils and waters, notably groundwater. Atmosphere is another important environmental compartment, and is the receptacle of some other polluting emissions generated by oil& gas activities. As in Chapter I, the spatial location of these emissions will be crucial for identifying where could be the impacted areas and the intensity of these impacts. Therefore, key infrastructures need to be considered and remain to be evaluated, namely the *oil separation batteries* and *flare stacks*.

Chapter II deals with atmospheric emissions released by burning of associated petroleum gas after crude oil is separated from water at oil separation batteries. Actually gas flaring occurs not at the oil separation batteries but at so-called flare stacks. These infrastructures are of different nature and usually are allocated at the oil separation batteries, but also where it is mostly needed, such as at places along the transportation and processing lines where pressure can prompt explosions. This is thus necessary to release the associated gas to the atmosphere for safety and security reasons. The impact of gas venting and flaring on overall degradation of environment and society is important. Therefore, as in Chapter I, emission estimates need to be calculated and mapped. Then in a second step their quality also need to be assessed. In chapter II, the **oilfields,** which are reservoirs located beneath the surface containing petroleum and associated gas, need to be identified and the potential quantity of gas to be released in the atmosphere needs to be calculated. The **flare stacks**, built-infrastructures placed on the surface, and therefore showing the spatial locations where gas will actually be released, are the focus of this chapter. The resulting spatial inventory maps attempt to provide enhanced representation, at high spatial scales and resolutions, of potential hazards impacting the atmosphere, lithosphere and hydrosphere.

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Chapter II - Spatial inventory of selected atmospheric emissions from oil industry in Ecuadorian Amazon: insights from comparisons among satellite and institutional datasets

Images of black carbon released at flare stacks in Shushufindi

Article submitted to *Safety Science*

Abstract

Atmospheric emissions from oil activities impact human health, socioeconomic status and exacerbate global warming. This study was conducted in the North-eastern Ecuadorian Amazon, a rich biodiverse and cultural area. This study aimed to show the benefits of public institutional data to advance hazard mapping knowledge for comprehensible risk evaluation. A spatial inventory was built from publicly disclosed reports spanning ten years (2003 to 2012). Emissions were estimated for gas flaring, associated black carbon (BC) and greenhouse gases (i.e., CO_2 and CH_4). To assess the quality of publicly available data, the calculated emissions were compared with satellite observations and historical energy statistics from the United Nations (UN). Results indicate total gas flared for this period of 7.6 Gm^3, corresponding to 760 $Mm^3.yr^{-1}$, and equivalent to a 3.7 – 4.5 $kt.yr^{-1}$ of BC. These values were in agreement with the UN estimates, suggesting that publicly available data are of acceptable quality. In contrast, the results from energy censuses diverged from satellite observation data, which might be explained by a poor calibration of satellite sensors. Study results enabled emissions mapping at a higher spatial scale than previous studies. Black carbon presented the highest results with 29.4 -148.0 $kg.m^{-2}.yr^{-1}$ in the cities of Shushufindi and Joya de Los Sachas. Greenhouse gases were up to twenty-fold higher than previous estimates. Publicly disclosed data estimates were discussed in terms of their potential on evaluations for climate, local health and economic impacts, to raise environmental monitoring and accountability in governmental institutions.

Keywords: black carbon, greenhouse emissions, public disclosure, spatial inventory, the Amazon

Highlights bullet points

- Publicly disclosed data were compared to UN official estimates and satellite observations.
- Estimates from public data were similar to those from UN, thus considered of acceptable quality.
- Inter-annual decline of gas flared according to public data suggests greater gas conservation.
- Airborne emissions with potential impact on health and climate were estimated.
- Spatially homogenized emissions represent a progress towards hazard mapping.

II.1. Introduction

The North-eastern Ecuadorian Amazon (NEA) is considered a worldwide biodiversity hotspot, encompassing a large number of endemic flora and fauna (Bass et al., 2010; Myers et al., 2000). Rich oil reserves have increased national economic growth, supporting improvements of education and health services in Ecuador (Larrea, 2006). However, local communities within the NEA allege negligible benefits, conversely enduring decades of related oil pollution, conveying health and environmental degradation (Baynard et al., 2013; Butt et al., 2013; Finer et al., 2008). Oil-related contamination has exacerbated the negative effects of human settlement including: decline in livestock health (Waldner et al., 2001); reduced yields in crops (Dung et al., 2008); reduced human health (Chang et al., 2014; San Sebastián et al., 2001; San Sebastián and Hurtig, 2005); degraded environmental amenities; reduced rural land values (Boxall et al., 2005); and reduced biodiversity richness (Finer et al., 2008; Jernelöv, 2010). These claims have resulted in twenty five years international legal challenges between local communities and Chevron (Buccina et al., 2013; San Sebastián and Hurtig, 2005). In 2018, an international court ruled in favor of Chevron, while the pollution remains in the NEA (BBC News, 2018).

Oil and gas extraction releases hazardous emissions attributed to several routine safety and security operations (Jernelöv, 2010). The atmosphere is affected by long-term venting and flaring of associated petroleum gas (APG) at separation batteries. Venting and flaring are necessary processes that occur when gas and water are separated from crude oil. They occur for pipeline security and safety reasons when means of transportation are lacking and re-injection to improve oil production in mature oil wells or electricity generation is not possible (Huang and Fu, 2016).

Venting and flaring produce atmospheric pollutants, comprising aerosol particulate matter (PM), including black carbon (BC), and greenhouse gases (GHG) such as methane or carbon dioxide. Gas flaring has the potential to acidify rainwater (Anejionu et al., 2015b). Black carbon (BC) is released during gas flaring and acts at a local level, resulting in soil calcination, degradation and destruction of vegetation (Solov, 2011). BC is a major light absorbing fraction of PM aerosols (Bond and Bergstrom, 2006) formed from incomplete combustion. Their radiative heating properties (Jacobson, 2001) transform it into a short-lived climate forcer with potential adverse effects on health (McEwen and Johnson, 2012) such as cardiopulmonary morbidity and mortality (Anejionu et al., 2015a; Giwa et al., 2014; Huang and Fu, 2016; Janssen et al., 2011). BC is classified as carcinogenic to humans by the International Agency for Research on Cancer (IARC, 2012). On the other hand, carbon dioxide (CO_2) and methane (CH_4) are climate change forcers with significant warming potentials (Soltanieh et al., 2016) and methane has energy generation capabilities, misused by venting or flaring (Anenberg et al., 2012; Boden et al., 2012; Dlugokencky, 2003; Robalino-López et al., 2014; Simpson et al., 2012). Besides, reduction of BC and methane emissions provide co-benefits in reducing specific vulnerability to heat waves, to droughts and flooding, to changes in the distribution of vector-borne diseases (Haines et al., 2006) and to agricultural losses (Shindell et al., 2012), since it improves air quality, economy, and human health altogether (Anenberg et al., 2012; Haines et al., 2006; Shindell et al., 2012).

Until 2006, Ecuador flared most of its APG (Peláez-Samaniego et al., 2007) resulting in energy and economic losses as well as atmospheric emissions to the environment (Ite and Ibok, 2013). The Ministry of Energy and Mines (MEM) and the Amazon Defense Front (ADF) have attempted to estimate these emissions in the NEA. Three distinct periods of management activities in Ecuador have been identified (Juteau et al., 2014):

- T1 (1972-1991): Foreign Texaco and National Petroleum Corporation (CEPE) engaged in oil activities;
- T2 (1992-2001): the State-owned Petro-Ecuador company assumed oil production and data compilation on gas produced was requisite;
- T3 (2001-2012): the State-owned Petro-Ecuador company assumed oil production and APG data reporting to Ministry of Environment (MAE) is mandatory.

Spatial gridding allocation of disaggregated emissions is useful for several purposes, such as pollution hotspot analysis, trajectory modelling, hazard and exposure mapping, multiple-pollutant and risk assessments (Andreo et al., 2006; Guttikunda and Calori, 2013; Lahr et al., 2010; Lahr and Kooistra, 2010). Oil emission sources and present perception on contamination potential have been spatially interpreted from local population surveys (Maestripieri and Saqalli, 2016) but need more detailed quantitative analysis. The present study was undertaken as a part of the trans-disciplinary MONOIL research program focused on environmental monitoring, health, society and oil in Ecuador (MONOIL, 2017).

This study first aimed to examine data quality acceptability, specifically determination on whether sufficient disclosure of energy data from the Ecuadorian government occurs, for the calculation of reliable estimates of APG flared at a regional scale. This was done using data extending over the T3 management period, supposedly characterized by improved monitoring (SENPLADES, 2013). In order to assess the available data's quality, our estimates were compared to United Nation data and to satellite observations provided by the National Oceanographic and Atmospheric Association (NOAA).

This study's second objective was to map airborne emissions using the Ecuadorian publicly available dataset, as long as their estimates were reasonably accurate to undertake mapping and to provide atmospheric local emissions at flare stacks. These evaluations and maps may support future hazard mapping and the improvement of security and safety planning.

II.2. Materials and Methods

II.2.1. Study Area

The study area (Fig. 12A) was restricted to the provinces of Sucumbíos and Orellana in the NEA (~144-900 m.a.s.l., Amazon lowlands), representing a 35,051 km^2 area (Fig. 12B). this area includes upstream and midstream oil and gas production infrastructure which are related to potentially polluting activities, likely to impact forests, rivers and streams but also cities, villages and extensive farming land. This study excluded rivers with high flow rates (i.e. Napo, Tiputini, Coca, Payamino, Putumayo, Cuyabeno and Aguarico) where the dynamic surface and groundwater interactions are poorly understood. The study area is characterized by a warm climate with a temperature range of 24°c to 35°c, and an average annual rainfall of 2,900 $mm.yr^{-1}$ (INAMHI, 2018). The hydrology regime is irregular with 1,000 to 5,000 $m^3.s^{-1}$ daily discharges, and characterized by flash floods, due to extreme sensitivity to rain events (Laraque et al., 2007). The Intertropical Convergence Zone (ITCZ) is responsible for complex atmospheric processes in the NEA, which is influenced by trade winds with changing direction throughout the year (Palermo and Parra, 2014).

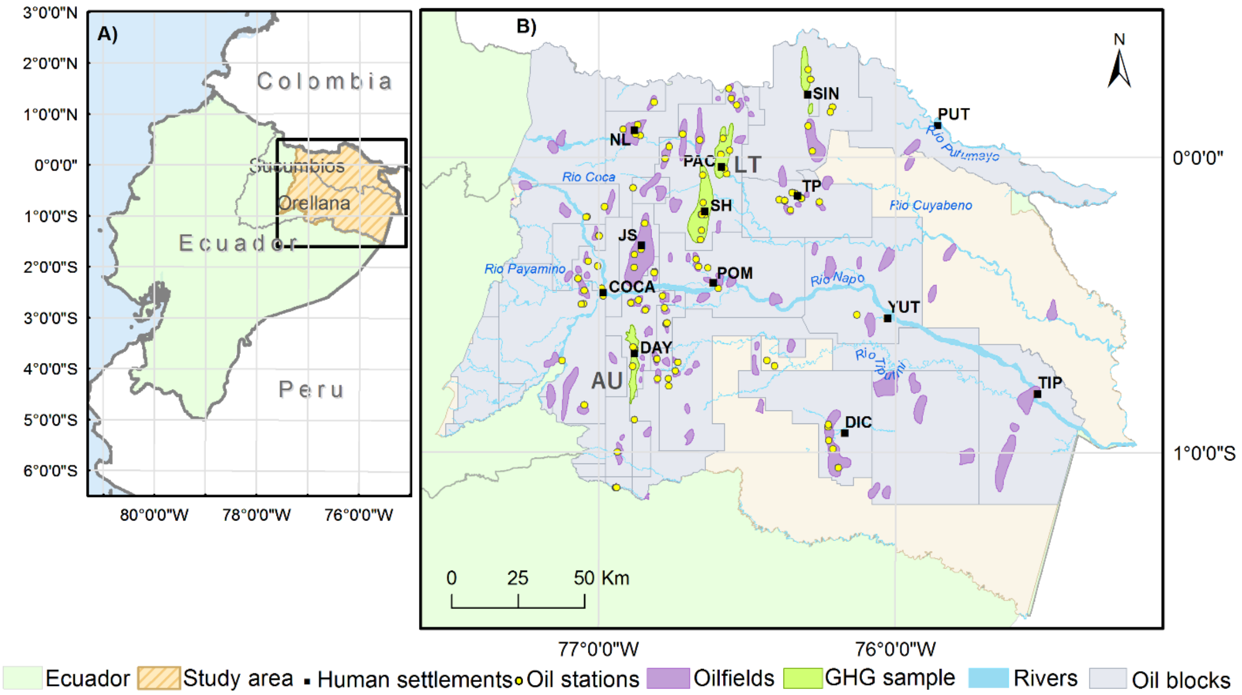


Figure 12. Location of flaring infrastructures in the NEA. (A) The North-eastern Ecuadorian Amazon (NEA) in Ecuador. (B) The study area within the two NEA provinces of Sucumbíos and Orellana including oilfields and oil stations. Represented human settlements are abbreviated: DAY= Dayuma; NL = Nueva Loja (aka Lago Agrio); SH = Shushufindi; TP = Tarapoa; PAC = Pacayacu; POM = Pompeya; PUT = Putumayo; COCA = Puerto Francisco de Orellana; JS = Joya de Los Sachas; YUT = Yuturi; DI = Dícaro; SIN = Singue; TP = Tiputini. GHG sample refers to greenhouse gases sampled oilfields, which are: SH = Shushufindi; LT = Libertador; AU = Auca; SIN = Singue.

II.2.2. Data for atmospheric emissions and comparisons

Energy bulletins and long-term reports of APG were compiled by the National Board of Hydrocarbons' website (NBH, 2018). Most of the historical energy reports are produced by the current Agency of Hydrocarbons Regulation Agency (ARCH) and the former National Hydrocarbons Directive (DNH) retrieved during compulsory environmental impact assessments. Several bulletins from the relatively new (created in 2008) Ministry of Strategic Sectors (MSS, 2015, 2014, 2013) were also integrated into several parts of the methodological process. These reports were disaggregated and categorized on a yearly basis for the 2003-2012 period. Institutional spatial data layers of oil infrastructure, including oil fields, flare stacks and stations, were obtained during the 2014-2017 partnership between ANR-MONOIL research Program and the Ecuadorian Ministry of Environment Program of Socio-environmental Remediation (MAE-PRAS). Data used in this study is summarized in Table 9.

Table 9. Database compilation used for airborne emission estimates and spatial allocation.

Spatial data	Description	Sources
Populations	Point	SENPLADES (2017)
Flare stacks	Point	NOAA (2015) MAE-PRAS (2014)
Oilfields	Polygons, 1:250.000	National Board of Hydrocarbons (2014) Petro-Ecuador (2013)
Refineries and batteries	Point	MAE-PRAS (1972-2014)
Non spatial data		
Associated gas	Gas exploited at oil fields	National Board of Hydrocarbons (2003-2014)
Gas composition	Molar fraction	National Board of Hydrocarbons (2016) Guerra del Hierro (2014)
Oil production	Records of monthly production	PetroEcuador (2001-2012)
Utilized gas	Historical data (2003-2012)	National Energetic Balance (2013) Ministry of Strategic Sectors Bulletins (2013-2014)
Transportation	Historical data (2007-2012)	National Institute of Statistics and Census (INEC)

Table 9 indicates the multiple datasets used for comparison of the results obtained in this study. Comparison analysis considered the materials, methods and type of analysis implemented to estimate airborne emissions. Other estimates of APG flared volumes were retrieved from night-time satellite observations of gas flares provided by the National Geophysical Data Centre (NOAA) and from historical energy statistics compiled by the United Nations (UNDATA). Thus, the emission factors (EFs) used in this study were applied to BC calculations. CO_2 estimates were directly retrieved from the Carbon Dioxide Information Analysis Center (CDIAC). The methods used in this study are described in section II.2.3.

Table 10. Datasets and calculations used for airborne emission comparisons (APG: associated petroleum gas; BC: black carbon; EF: emission factor).

Data source	Material	Type of analysis	Pollutant estimated
NOAA	Satellite observations (2003-2012)	Estimated APG flared was multiplied by EFs used for this study	BC
UNDATA	Historical energy statistics (2003-2012	Estimated APG flared was multiplied by EFs used for this study	BC
CDIAC	Historical energy statistics (2003-2012)	Estimates retrieved were directly used	CO_2
This study	See table 9	Production equations and upper and lower EFs.	BC
		Production and mass balance equations for carbon dioxide and methane.	CO_2, CH_4

II.2.3. Emission processing and calculations

II.2.3.1. Gas flaring

In Ecuador, data on flared or vented gas emissions are not readily available. Institutional technical reports and bulletins were analyzed for T2 and T3 periods to facilitate a description of the technology and infrastructure implemented and to derive emission estimates. In general, the analysis followed the upstream production process, which includes extraction, transportation to refining and final export (Foss, 2012). Except for the proportion of gas leaked or used, APG is assumed to be flared. Data is disaggregated on an annual basis to calculate potentially flared gas at each oilfield.

The general procedure to obtain total gas flared is described in Equation 3:

$$Gas\ flared(m^3) = \sum_{i=1}^{50} \mathrm{APGnu}_i - (\mathrm{APGe}_i + \mathrm{APGf}_\mathrm{i}) \qquad \text{(Eq.3)}$$

Where,

i = the index of the oilfield considered (50 oilfields in total)

APG*nu* = unused APG in each single oilfield.

APG*e* = input for electricity generation at Shushufindi Gas Plant, only reported for years 2005 to 2012, and representing 8.3% of total APG_{nu} (MSS, 2014, 2013)

APG*f* = leaked fugitive emissions, representing 3% of total APG_{nu} (Larsen et al., 2015; MSS, 2013).

II.2.3.2. Black carbon

BC emissions were obtained using emission factors (EFs) which were measured from the heating capacity of gas burning at flare stacks (McEwen and Johnson, 2012). In Ecuador, no official EFs for BC have been defined, hence estimates were calculated using the regional baseline emission factors (EF = 0.5) and upper bound values (EF = 0.75) (Huang and Fu, 2016; McEwen and Johnson, 2012), following Equation 4 (Giwa et al., 2014):

$$BC(t) = EF\ \left(\frac{t\ of\ BC}{10^3 m^3}\right)\ x\ \mathrm{Vol.gas\ flared}\ (10^3\mathrm{m}^3) \qquad \text{(Eq.4)}$$

In addition, BC emissions were calculated from gas flaring estimates from the United Nations at the national level (which was relevant as virtually all gas flaring in Ecuador occurs within the study area), and from the National Geophysical Data Centre (NOAA-NGDC) long-term inventories of satellite products, in order to compare with our estimates. Emissions were expressed as the sum of emissions per oilfield, average emissions per single oil-field and flaring infrastructure, i.e. flare stacks and batteries, and total emissions for the study area.

II.2.4. Atmospheric emission mapping

Individual and battery-associated flare stacks identified from NOAA satellite data and MAE-PRAS were regrouped in a single spatial layer. Total gas flared in each oilfield, computed as described in section II.2.2.1, was divided equally to corresponding flaring points following a four-step hierarchical approach:

(1) if a flare stack was inside an oilfield, it was assigned directly to its corresponding oilfield (n = 143);

(2) if a flare stack was outside oilfields, it was attributed to oilfield by nearest distance method (n=38). Some flares were not found to occur with a specific oilfield (1) or occur in close proximity to an oilfield (2). In this case,

(3) flared gas was attributed to north-western flare stacks based on the production process from reservoir to export (Foss, 2012). This indicates APG flows from secondary pipelines to north-western Ecuador, connecting with processing facilities to two core pipelines: Pipeline System of Trans Ecuadorian (SOTE) and Heavy Crudes Pipeline (OCP).

(4) Fictitious flaring stacks were allocated to oilfields lacking flaring infrastructure but with registered APG and crude oil production.

The number of fictitious flare stacks added to each oilfield would ideally have been estimated through the correlation between number of flare stacks and quantity of APG. However, the weak coefficient of determination (r^2 = 0.22) hindered the estimation of the number of flare stacks to be placed within or in proximity to an oilfield, hence a centroid was created for each oilfield, as often implemented in spatial gridding allocation studies (Benkovitz et al., 1996; Rehr, Amanda P; Small, Mitchel J; Mathews Scott H; Hendrickson, 2010). In every aforementioned case, the assignation of BC emissions to a single flare stack was expressed using total min-max BC values at each oilfield divided equally to the corresponding number of flare stacks.

A cell-grid layer was created to sum-up BC emission of each flaring facility. Cell size can be determined according to the study area size and spacing of observed emissions (Bertazzon et al., 2014). The spacing between single flare stacks was considered too large to use a symbol to represent quantity. Representation of emissions was considered to be more informative when presented in 25 km^2 grid-cell for the time period considered (also mapped in 9 km^2 and 16 km^2

grid-cells, but not shown; and the data can be rescaled to 1x1 km also as needed to input in transport models). The grid-cell size seemed well adapted to the spatial scale considered in this study, as dispersion plumes have previously been simulated in regional models such as the CCATT-BRAMS (in the Brazilian Amazon) using 15 and 30 km^2 grid-cells (Freitas et al., 2009) and global models such as the MEGAN V.2.1. of 3,025 km^2 grid-cells (Sindelarova et al., 2014). Besides, this spatial resolution could be evaluated in terms of cumulative, added or synergistic effects with other hazardous substances (Lahr et al., 2010), i.e., particulate aerosols (PM 2.5,10), spilled hydrocarbons, oilpit brines that could probably be within a similar or lower dispersion ranges, also analyzed within the framework of the trans-disciplinary ANR-MONOIL research program (MONOIL, 2017). Spatial treatments were performed in ArcGIS® system. Values were mapped via Jenk natural breaks.

II.2.5. Carbon dioxide and methane

CO_2 and CH_4 were calculated using the Intergovernmental Panel on Climate Change (IPCC) - Tier 2 method which denotes a combination of production and mass balance equations (IPCC, 2006a). The calculation procedure incorporated several data on chemical properties of gases, as shown in Equation 5 and 6. Estimates were made for four major oilfields because data was complete and available (n = 22 oil wells in total): Auca, Shushufindi, Singue and Libertador, see Fig 1B.

$$E_{CO2,oilprod,flaring} = GOR \; x \; Q_{oil} \, (1 - CE) \, X_{flared} \, M_{co2} \, [y_{co2} + (Nc_{CH4} \; x \; y_{CH4} + Nc_{NMVOC} \; x \; y_{NMVOC})(1 - X_{soot}) \,] \; x \; kmol \qquad \text{(Eq. 5)}$$

$$E_{CH4,oilprod,flaring} = GOR \; x \; Q_{oil} \, (1 - CE) \, X_{flared} \, (1 - FE) \; x \; M_{CH4} \; x \; kmol \qquad \text{(Eq. 6)}$$

Where,

$E_{k,\,oil\,prod,\,flaring}$ = direct amount ($Gg.yr^{-1}$) of GHG *k* emitted due to flaring at oil production facilities.
GOR = average gas to oil ratio ($m^3.m^{-3}$) referenced at 15°C and 101.325 kPa.
*Q*oil = total annual oil production ($10^3.m^3.y^{-1}$).
Mgas = molecular weight of gas of interest (e.g. 16.043 for CH_4 and 44.011 for CO_2).
NMVOC = non-methane volatile organic compounds.

Nc_i = number of moles of carbon per mole of compound (e.g. 1 for both CH_4 and CO_2, 4.6 for NMVOC).

y_i = mol or volume fraction of the associated gas that is composed of substance.

CE = gas conservation efficiency factor.

X_{flared} = fraction of the waste gas that is flared, using estimates from UNDATA.

FE =flaring destruction efficiency (i.e., fraction of the gas that leaves the flare partially or fully burnt, typical a value of 0.98 assumed for flares at production and processing facilities).
X_{soot} = fraction of non CO_2 carbon in the input waste gas stream that is converted to soot or particulate matter during flaring.

kmol = is the number of kmol per m^3 of gas referenced at 101.325 kPa and 15°C (i.e. 42.3 x 10^{-3} $kmol.m^{-3}$ for CH_4 and 4.23 x 10^{-3} $kmol.m^{-3}$ for CO_2) multiplied by a unit conversion factor of 10^{-3} $Gg.t^{-1}$ which brings the result of each applicable equation to units of $Gg.y^{-1}$.

II.2.5.1. Estimating variations based on percentile change in key parameters

The required data to implement Equation. 5 and 6 should be site specific, however the IPCC guidelines propose default values if no site specific information is available. All input parameters could be obtained or defined using suggested defaults, except the conservation efficiency (CE) and Xflared (IPCC, 2006a). The CE factor expresses the amount of produced gas and vapour that is used for fuel, produced into gas gathering systems or re-injected, using a value of +1 if all gas is conserved and value of 0 if all gas is flared. CE values may fluctuate between these two values depending on the technology implemented at a specific site, while Xflared expresses the fraction of gas that is rather flared than vented (IPCC, 2006a).

Sensitivity analysis is recommended for variables to prioritize efforts to develop good estimates (IPCC, 2006b). CE and Xflared depend on expert knowledge-based assumptions on the type of technology implemented. A simple sensitivity analysis through percentile variation - in 10% increments (Sala et al., 2000; Sax and Isakov, 2003) was performed for CE and Xflared. The range of values were calculated for the potential minimum (10% flared 90% conserved) to the maximum values (90% all gas is flared 10% conserved), assuming values of less than 10% for gas flared to be an unlikely outcome because at least 8.3% APG is used for energy generation (MSS, 2014, 2013) and 3% APG is leaked (Larsen et al., 2015). Likewise, the gas flared fraction used as input parameter was also varied in percentiles from 10^{th} to 90^{th}, since these values are more likely to occur. Subsequently, a range of possible min and max values for every oilfield in year-on-year basis were created using this approach to Xflared and CE values.

II.3. Results

II.3.1. Gas flaring and black carbon emissions according to publicly available data

Gas flares for the T3 period totaled 7.6 Gm^3 for 53% of oilfields (n = 50 of 94). Estimated BC emissions during 2003-2012 ranged from 3.7 to 4.5 Mt. Fifty oil fields had APG data associated with them and eleven did not have flare stacks. Fictitious stacks were allocated (centroids) to oilfields where no flare stack was evident. At least, 52.8% of APG (4.3 Gm^3) was flared at four out of fifty oilfields operated in the NEA: Shushufindi, la Joya de Los Sachas, Libertador, south from Putumayo River and Yuturi. Estimates were described using median (interquartile range). The gas burnt per oilfield was 39.3 Mm^3 (5.7-100 Mm^3) and 15.2 Mm^3 (5.7-35.1 Mm^3) per flare stack (n = 181). All selected emissions, including GHG were displayed in Table 11. These allows a comparison between all selected hazardous emissions, simultaneously. Resulting GHG emissions were explained in section II.3.3.

II.3.2. Estimates of this study compared to other datasets

UNDATA emissions, although estimated at the national level, reported fluctuating estimates on a year-on-year basis compared to this study from 2004 to 2005 and from 2009 to 2012. Overall, UNDATA estimates were very close to this study's estimates from publicly available data, and those were highly correlated (r = 0.84). Thus, to effectively compare data from satellite observations to other datasets, an average curve for APG flared and BC was calculated from historical energy statistics data. All these institutional datasets are referred hereafter as Energy Census Data (ECD).

Figure 13 shows that calculated emissions from NOAA satellite data observations contrasted with ECD average estimates. Annual variations (2003-2012) indicate a widening gap from 1.2 to 2.5 fold between satellite observations and ECD average, showcasing the strong negative correlation between the two datasets (r = –0.90). According to this this study the highest BC emissions occurred in 2003 (516 t). Emissions in subsequent years decreased to reach 401 t in 2012.

The higher NOAA satellite estimates indicate an increasing trend, reaching up to 945 t in 2012. Overall, gas flaring and BC emissions were weakly correlated to oil production (Figure 13).

Annual oil production seemed to be globally stable during T3, with year-to-year fluctuations between 17 and 22 Gm3.

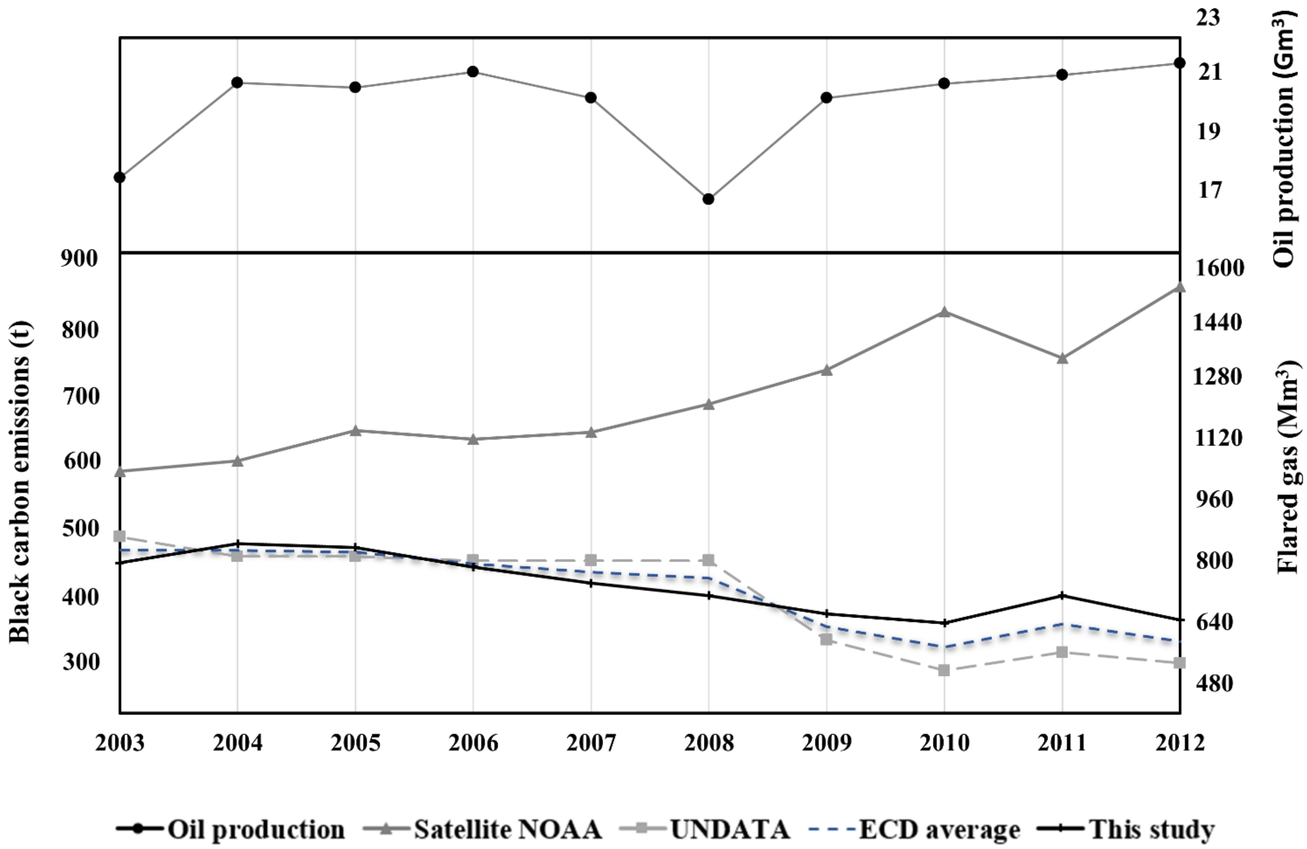


Figure 13. Temporal patterns of flared gas volumes and corresponding BC emissions in tons. Total oil production from all Ecuadorian fields is also plotted for comparison. Correlation coefficients between flared gas and oil produced were: r = 0.39 for NOAA satellite data; r = - 0.30 for ECD average. Data is disaggregated for oil production from Bulletins of the National Board of Hydrocarbons' website, by oil field, restricted to the study area.

II.3.3. Mapping of airborne black carbon emissions at a regional scale

BC estimates ranged between 2.7 - 1,153.5 $kg.km^{-2}.yr^{-1}$ (lower EF) and 4 – 1,730 (upper EF) across grid cells. The visual display was the same for both EF because the values are proportional. Highest emissions were located at Libertador oilfield, crossing north to south from Shushufindi to Pompeya, west to east from Joya de Los Sachas to Yuturi oilfield, and at Dícaro. Medium to low emissions were found from Tarapoa to Putumayo (Figure 14B).

Table 11 Total and annual median gas flared and BC emissions in the T3 managerial period. BC is expressed in tons with base (EF=0.5) and upper bound (EF=0.75) emission factors. Average (±SD) values for GHG estimates. Only the four highest oilfields are shown (gas flare and BC). Flare stack values exclude oilfields with no declared flare stacks.

Oilfields (n = 50)	Gas flared		BC		CO_2		CH_4		
	Total (Mm^3)	Median [d] ($Mm^3.yr^{-1}$)	Total (t)	Min-Max ($t.yr^{-1}$)	Total (kt)	Average ($kt.yr^{-1}$)	Total (kt)	Average ($kt.yr^{-1}$)	Contribution[b] (%)
Shushufindi (SH)	1,479.2	147.7 (0-176)	646-891	80-121	53,289.67	5,921.1± 756	474.28	158 ± 15	17%
Libertador (LT)[c]	928.6	92.4 (50-109)	378-500	48-62	24,586.10	2,731.8 ± 1,428	318.61	106.2 ± 11	12%
Sacha (SC)	859.1	83.2 (75-92)	340-496	42-61	-	-		-	11%
Yuturi (YT)	704.8	74.6 (23-78)	277-469	32-58	-	-		-	9%
Auca	213.0	17.3 (13-21)	89-111	11-14	9,874.57	1,097.2 ±1,775	193.24	64.4 ± 5	3%
Singue	71.1	6.3 (1-10)	31-43	4-5	2,968.46	329.8 ±201	22.15	7.4 ± 0.5	1%
Single oilfield	39.3 (5.7-100)	4.1 (0.4-13)	62-91	7-10	4,833	537.1	17.9	2	-
Flare stacks	15.17 (5.7-35.1)	1.89 (0.7-4.4)	190-278	5.1-7.5	1,362	151.4	5.1	0.6	-
Total T3	7,600	782 (732-816)	3,735-4,583	373-572	82,144	6,106 ±4.6	304.64	25.3 ± 36.4	100%

[a] GHG estimations for total NEA based on key (n= 4; 36.8% of total) sampled oilfields. Totals were calculated using a 2.72 factor.

[b] The contribution column is calculated based on gas flared and BC only.

[c] Corresponds to a group of 5 oilfields where GHG emissions are calculated using data from top flaring oilfield: Pacayacu.

[d] Median and interquartile ranges at 25th and 75th percentiles.

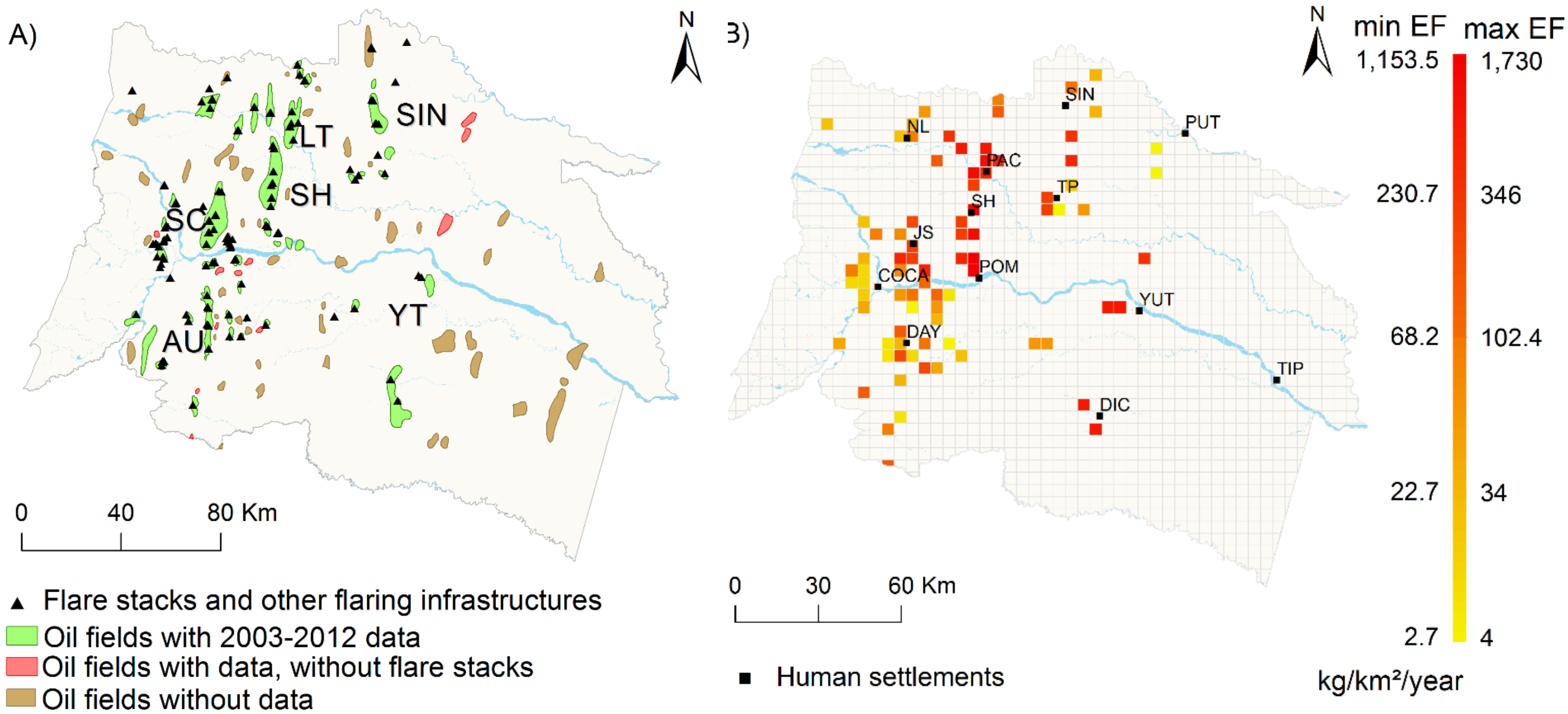


Figure 14. Spatial variations of airborne emissions within the North-eastern Ecuadorian Amazon (NEA) from 2003 to 2012. A) Air pollution related infrastructures: flare stacks, batteries or refineries and main oilfields (AU: Auca; SH: Shushufindi, SC: Sacha, SIN: Singue; LT: Libertador, YT: Yuturi). B) Black Carbon emissions reported at upper and lower bound average values. The values scale show lower and upper EF emissions.

II.3.4. Greenhouse gas estimates

Carbon dioxide and methane emissions were estimated for three years, in four out of fifty oilfields with reported APG. Figure 14A indicates where these oilfields are located. Figure 15 and Table 12 show total and average GHG emissions for three representative years per oilfield. Variability across oil fields was slightly higher for methane emissions than carbon dioxide. Annual average reported per flare stack (country-specific data) accounted for 271 ± 98 $kt.CO_2.yr^{-1}$ and 1 ± 0.7 $kt.CH_4\ yr^{-1}$ within the oilfields sampled.

This estimate represents 36.8% of the total GHG emissions resulting from APG flared. These estimates should thus be multiplied by a factor of 2.72, to obtain GHG corresponding approximately to the total APG flared for the entire NEA in T3. Therefore, the NEA would totalize 82.1 $MtCO_2$ and 304.6 $ktCH_4$ during this period, equivalent to an annual average of 6.1 $Mt.yr^{-1}$ CO_2 and 25.3 $kt.yr^{-1}$ CH_4 (Table 12). Figure 15 presents estimates and variations for the four oilfields.

GHG estimates can only be compared to other datasets for CO_2, because CH_4 has not been previously evaluated in the NEA. CO_2 has been previously estimated in a disaggregated manner by the Carbon Dioxide Information Analysis Center (CDIAC) enabling gas flaring to be compared only for years 2003, 2007 and 2012. The approach used in this study showed twentyfold higher emissions than CDIAC estimates.

Table 12. Total and average annual GHG emissions for four oilfields from sampled data collected in 2003, 2007 and 2012 (n =22 oil wells).

Oilfield	CO_2		CH_4	
	Total (Mt)	Annual average (±SD) ($Mt.yr^{-1}$)	Total (kt)	Annual average (±SD) ($kt.yr^{-1}$)
Auca	3.3	1 ± 0.7	21.5	7 ± 5
Libertador	8.2	2,731 ±1.4	35.4	12 ±11
Shushufindi	17.8	5,921 ±1.8	52.7	18 ±15
Singue	0.88	329 ±0.2	2.5	0.8±0.5
Total (n=4)	30.2	2,245±0.5a	112	9.3±4.5a

[a] ±SD calculated using pooled estimators. Rounded data.

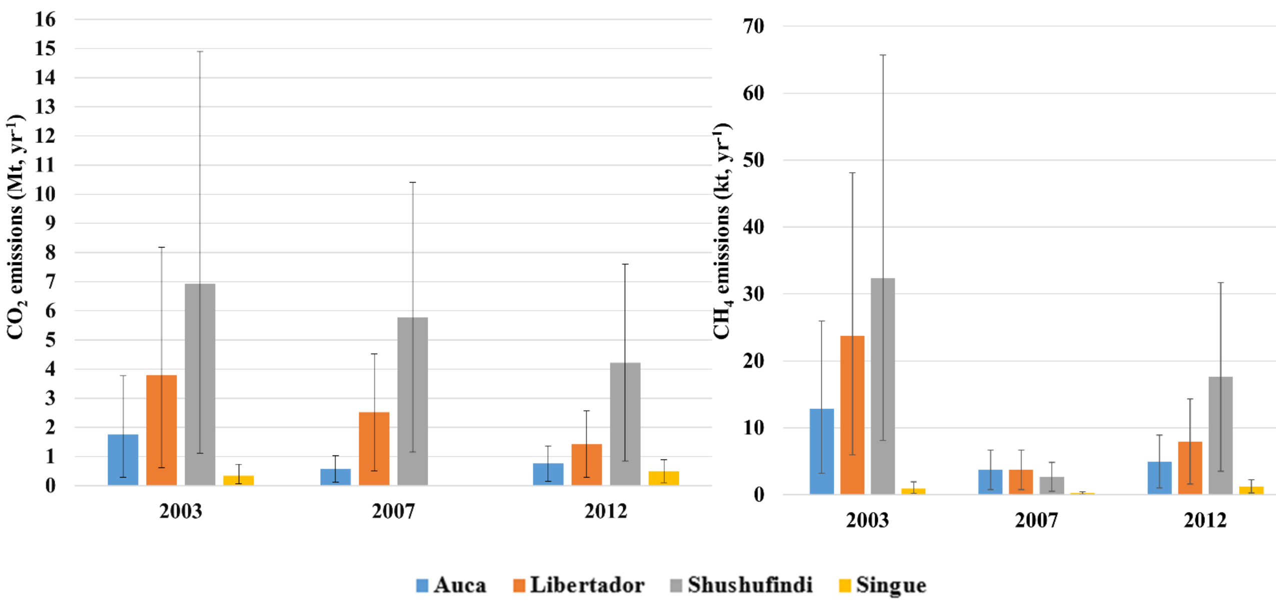


Figure 15. Coloured bars illustrate CH_4 and CO_2 estimates with percentiles of CE and Xflared at 0.5. Error bars represent estimate variation with percentiles of CE and Xflared between 0.1 and 0.9. Estimates were derived for 3 years and 4 oil fields where data were complete.

II.4. Discussion

II.4.1. Data reporting

Higher levels of disclosure, and improvement in statistical analysis and collection methods may increase inventory reliability and value in supporting safety and security decisions. Public disclosure benefits environmental accounting. Constraints in information access hampers proper estimations. This is especially true in oil industry, whose accidental emissions or discharges are seldom reported (Jernelöv, 2010). In the NEA, APG data reported for 2005 and 2008 is incomplete for some oilfields, e.g. APG has not even been reported for Yasuní-ITT in Fig. 2., a famous oil field for its conservation initiative aiming in keeping oil underground (Larrea and Warnars, 2009; Vallejo et al., 2015). Similarly, there is uncomplete and limited data available for flaring infrastructure in some oilfields, such as Vinta, Yuca Sur or Pañacocha.

Ecuador had signed the Kyoto Protocol (1992) but during T3 had no flaring and venting targeted policies (WorldBank, 2004). The protocol established that regulating institutions and producing companies should be independent entities. Both tasks were in charge by the same institution in Ecuador. In most developing countries, institutional responsibilities for gas flaring and venting are often nontransparent, conflicting, and ineffective (WorldBank, 2004). This lesson was later recognized and, in 2008, institutional changes were made (Rwengabo, 2018), which is probably reflected by the sudden drop in APG flared from 2008 onwards, as shown by the ECD trend lines in Figure 13. This suggests that the drop in APG flared would be related more to an increase of gas use efficiency than to a moderate decrease in data disclosure. It is possible that some quantities of APG were unreported, as the satellite calibration problems at that time (see section II.4.2.2.) obviously not explain the full gap between estimates from the NOAA and from energy census data. One third of the data in our study comes from open government data, which is well in conformity with the Ecuadorian Transparency and Access to Public Information Law (LOTAIP). Ecuador ranked 58th out of 93 countries in e-government data disclosure and open data, especially for airborne pollutant emissions (Open Knowledge International, 2015) and has 44% of open data implementation, indicating an important level of disclosure from public institutions, still this is 10% less than the South American average (Triviño 2016). Other countries improve disclosure as a way to better control their emissions, which is an important step to meet zero-emission goals defined within the Global Gas Flaring Reduction Partnership (GFFR) (World Bank Group, 2018), e.g. Nigeria openly reports total gas flared (Giwa et al., 2014). The Ecuadorian government does not formally belong to this partnership (World Bank Group, 2018) but the state company has engaged in voluntarily attempts to meet these targets.

II.4.2. Emission estimates

II.4.2.1. Black carbon

In Ecuador, EFs and standardized measurements for heating values are in infancy stages. Upper bound EF had previously been extrapolated from Venezuela, assuming similar technology was implemented (Huang and Fu, 2016), and lower EF are also based on general estimates from heat values emissions (McEwen and Johnson, 2012). BC emission is an important fraction of PM with global warming and cancerigenous properties (Bond et al., 2004; Conrad and Johnson, 2015; De Miranda et al., 2012). Most recently, Conrad and Johnson, (2017), measured EFs for BC at four single flare stacks in Libertador oil field. The result they obtained was that BC emission rates are highly variable and underestimated worldwide. Using the resulted EFs, average emissions in the NEA were estimated at 8.3 – 23.9 t. $BC.yr^{-1}$ per flare stack. These values were slightly higher than our 5.1 – 7.5 t. $BC.yr^{-1}$ estimates, using reference EFs described in section II.2.3.2. These differences highlight the importance of increasing the sampling effort to properly define average EFs for the study area.

II.4.2.2. Greenhouse gases

The site-specific databases and derived estimates seemed to be a representative fraction of emissions from the oil sector. CO_2 emission estimates in this study were up to twenty fold of those estimated by the CDIAC (Boden et al., 2012). To obtain similar yields estimated in this study, CE (conservation efficiency) and Xflared (fraction of flared APG) values need to be set at 90% and 30% respectively. Those values are unlikely to occur considering the used and leaked APG fractions. On the other hand, Ecuador is ranked 49th in CH (emissions from fossil fuel use. However, an annual ranking may not be the relevant way to present inter-countries comparisons as

Ecuador reports illustrate a high inter-annual variability in APG production (Simpson et al., 2012); integrating longer term data (e.g. 2-4 years) would probably be better.

II.4.3. Comparison of emission sources

II.4.3.1. Single flare stacks from various countries

Estimates at single flaring facilities indicate Ecuador, specifically the study area, remained a low global contributor to gas flaring for the period considered, with a total amount of 7.6 Gm^3 during the T3 managerial period, equivalent to an average of 4.19 Mm^3.yr-1 per flare stack. Comparatively, in the Niger Delta, 350 Gm3 of total APG was flared for a similar period (2000-2013), corresponding to 80 Mm^3.yr-1 for each single flare stack (Anejionu et al., 2015a). In Alberta, the median value for individual stacks reported was 0.03 Mm^3.yr-1 APG flared (Johnson and Coderre, 2011). The reported amount of flared APG per flare stack thus differs significantly, i.e., nine times lower emissions in Alberta than in the study area, and 19.5 times higher in Nigeria.

II.4.3.2. Institutional and satellite datasets

Instrument calibrations, data collection or calculations can result in errors portraying under or overestimation of emissions. For instance, estimates from the NOAA satellite sensor, prior to 2012, might reflect similar spectral responses with other important sources, i.e., cloud coverage, biomass burning, and be prone to overestimates, compared to the previous DMSP-OLS calibration technique. Approximately 25% higher APG flared was estimated from satellite data in Nigeria and in Bolivia, where overestimates were due to biomass burning (Anejionu et al., 2015b). In subsequent years (2013, 2014 and 2015) the superior, calibrated sensor, VIIRS, indicated 800, 1,000 and 1,100 Mm^3 gas flared respectively in Ecuador (World Bank Group, 2018), to be compared to the 1,500 Mm^3 in 2012 according to the former sensor (this study). This suggests an overestimation until 2012 for Ecuador as well, although it was not possible to assess its extent to provide a correction for the satellite imagery estimates. Conversely, the lowest UNDATA estimates (particularly for the 2009-2012 time period) consider gas flaring at the national level, possibly underestimating emissions due to missing discharges. Comparative analysis of airborne emissions suggests using several data sources is an asset in the evaluation of hazardous emissions. Long term estimates by satellite observations have also been prone to errors (Elvidge et al., 2009). Historical energy data and other methods of estimation are suitable for determining the activity sector specific emissions and for comparative analysis with independent non-governmental and satellite product reported estimations.

II.4.3.3. Across activity sectors

Oil production is key contributor of BC emissions in Ecuador, yet other potentially important anthropogenic emissions cannot be neglected, i.e., biomass burning (Fearnside, 2000; Ramanathan and Carmichael, 2008) and vehicle fossil fuel combustion (Gramsch et al., 2013). Data was insufficient to compare gas flaring with biomass burning, alleged to be an important source of contaminants (sulphur, nitrogen, carbon, and metals), due to long trajectories and depositions within the Ecuadorian Amazon (Barraza et al., 2018; Fabian et al., 2005). Nonetheless, available reports from the National Institute of Statistics and Census of Ecuador (INEC) enable comparisons with transportation emissions in the NEA. INEC reported the number of diesel vehicles ($\sum$ = 6,780 buses in 2012) has increased during 2007-2012 (INEC, 2017). With an estimated EF for diesel vehicles of 0.15 t.BC.bus^{-1}. yr^{-1} (Johnson et al., 2015) this results in an average of 264 ± 64 t.BC.yr^{-1} discharged during T3 period. These emissions arise from 20.5% of total vehicles. For the remaining 79.5%, corresponding to gasoline vehicles, a 0.03-fold factor may be applied to the emissions from diesel vehicles (Streets et al., 2001). Average vehicle emissions account for 485 ± 76 t.BC.yr^{-1}, comparable to 388 - 568 t.BC.yr^{-1} from flaring APG. In fact, this is in agreement with a local study which suggests that the concentrations of particulate aerosols from flared APG may be similar to urban sites (Barraza et al., 2018).

II.4.4. Potential economic, health, and environmental losses

Direct economic losses associated to gas flaring represent potentially important economic revenues for Ecuador. Considering our total estimates, and the conversion factor from thermal units (a unit commonly used for calculating the monetary value to generate electricity) of 0.24 US$.m^3 if sold in the United Sates (US. Department of Energy, 2017), the main potential Ecuadorian commercial partner, would result in a US$182.4 million.yr^{-1} economic losses. A partial estimate that should integrate monetary values of environmental and health costs, yet to be quantified. Inhalation is the main exposure pathway for local populations to airborne pollution in the NEA (Barraza et al., 2018). Black carbon is a carcinogen (IARC, 2012), having also short-term health effects (Janssen et al., 2011), and may serve as a carrier for toxic chemicals passing through the blood system, and cause cardiorespiratory diseases (Janssen et al., 2011). Hence, forthcoming hydrocarbon regulations could include BC and methane emissions.

II.5. Conclusion

This study presents a “bottom-up approach” to process and visually represent APG datasets in gridded form for specific activities (i.e. type of infrastructure) in oil and gas production of selected pollutants. Emission estimates and subsequent homogenized spatial distribution of pollutants in the NEA are needed for future transport modelling and hazard mapping. Willingness of institutions and operators to disclose data is key to enhance security and safety decision making, towards improved health, economic, and environmental conditions. During this study period (2003-2012) Ecuadorian institutions started to implement recovery technology. Institution reports and bulletins do not always provide appropriate data detailed or accurate data for emission estimation. This study has proposed an emission estimation approach that uses government data rather than the variable institutional datasets, and relies less on satellite products. Having several data sources for estimation is found to be useful. Finally, public access to gas flaring data from Ecuadorian institutions was found to be within acceptable levels (two-fold lower estimates than satellite observations), which is an important step towards accomplishing environmental monitoring and accountability objectives.

Future research aims to combine maps on oil spills and BC to address cumulative impacts, including other potential pollutants (e.g. total hydrocarbons, heavy metals) and source types (i.e. mud drilling pits). Hazard maps could assist decision making, when overlapping environmental (e.g. biodiversity values, surface waters, groundwater, etc.) and socioeconomic (poverty, medical service access, education, etc.) vulnerability maps. This study is a comprehensive first step to provide emission maps that could be used as input to assess atmospheric dispersion together with previous inventories and subsequently overlaid with vulnerability maps.

From oil spills and airborne emissions to vulnerability of biodiversity and natural heritage

Previous chapters I and II attempted to provide spatial emissions of pollutants, potentially impacting the lithosphere, hydrosphere and atmosphere. They have been estimated and subsequently homogenized when pertinent. Reporting and data quality have been discussed in terms of potential use for input to future spatial distribution models of pollutants in the NEA that surely need to be developed to entail more spatially fine-tuned hazard maps. Chapter II proposed a gas flaring process approach that provided estimates from government data rather differing amongst institutions datasets, including satellite products. It seems, following evaluations of atmospheric and spill emissions that the former are easier to assess for quality of data than oil spills.

Ecuadorian institutions have promoted the reduction of APG flaring for economic and environmental reasons, from enforced regulation on oil operators (ARCH, 2018). Nonetheless, local communities and environmentalist groups express concerns for even higher public disclosure and standards for better protection of ecological assets and the natural heritage it encompasses. Institution reports and bulletins were not always uniform regarding the details and the accuracy of data. This study suggests acceptable levels of data disclosure by Ecuadorian institutions and of public accessibility to these data, in light of environmental monitoring and accountability. This means that even if data quality may remain debatable, mapping emissions at this level is necessary, not only because of the populations that may be harmed, but also to assess environmental impacts on the exuberant biodiversity and ecological processes that are inherent to the Nigerian rainforests (Anejionu et al., 2015b; Ite et al., 2013) and in Amazon rainforests (Arellano et al., 2017) where oil activities are carried out routinely.

In next chapter III, the ecological vulnerability of natural heritage, including biodiversity, is assessed in a broad manner and at a regional scale, in the North-eastern Ecuadorian Amazon. To evaluate the intrinsic vulnerability of natural resources and biodiversity, seemingly the most outstanding environmental assets to protect in the NEA, one does need to evaluate biodiversity by measuring multiple taxa *"in situ*", which is probably a daunting task. In this context, the relevance of land use as a surrogate for biodiversity will be assessed, while protected areas will be used as a surrogate of ecological integrity. The practical purpose of this research should be to provide operational tools to address complex environmental problems via a systemic approach. The development of a vulnerability map for natural resource conservation planning is the ultimate goal, in an area where anthropogenic pressures, notably related to oil & gas activities, constitute a threat for natural heritage.

Chapter III - Vulnerability assessment of natural heritage and biodiversity in the North-eastern Ecuadorian Amazon using land use cover and nature protection status.

"Either we leave our descendants an endowment of zero poverty, zero fossil-fuel use, and zero biodiversity loss, or we leave them facing a tax bill from Earth that could wipe them out".

—Johan Rockstrom

Article *in preparation*

Abstract

This study examined the possibility to use publicly available data to assess the potential ecological vulnerability in terms of natural heritage and biodiversity value in the Northeastern Ecuadorian Amazon. For this purpose, current status of protection (2018) and most recent land cover (2015) were used as proxy indicators, respectively. One can hypothesize that areas with a high protection statute, suggesting an important natural heritage, and/or with a high biodiversity are more threatened of losing this heritage when facing any hazard related, for instance, to economic activities and are then considered as more vulnerable to such hazards. Present research includes the assignation of vulnerability rating scores to two types of attributes: (i) the status of protection is used to locate natural heritage sites and to rank these sites in terms of vulnerability, and (ii) land use is used to infer biodiversity value. A robust relationship was found between land use classes and species richness both for vascular plants in new world tropical regions and for multi-taxa in worldwide tropical forests. Spatial overlays and correlation analysis were performed upon output maps to search for spatial congruence between the two maps obtained. Results show that assigning values to vulnerability is possible. Primary forest has a significantly higher biodiverse value and therefore was the most vulnerable class. Other land use classes were also ranked, but their ranking reliability level was not conclusive. Several causes for this were discussed, e.g., border effect or proximity to primary forests. Overall, according to different taxa comparisons with the Kruskall-Wallis and Spearman correlation tests, biodiversity value assignation and spatial vulnerability indexing based on natural heritage are coherent. Medium spatial correlation (r =0.52) may be explained in part by 3,060 km^2 of non-protected primary forest. Finally, the overlay-index method for land use and protected areas proved to be suitable for building ecological vulnerability indexes at a regional scale.

Keywords: land use, protected areas, biodiversity value, conservation, spatial analysis

III.1. Introduction

Vulnerability is defined as the degree to which an exposed system or sub-system is sensitive to damage due to the pressure by a hazard or stressor (Gleyze, 2002; Metzger et al., 2006; Turner et al., 2003). The ecological vulnerability is defined as the potential of an ecosystem to regulate its response to a stress, occurring at various time and space scales and including also the common components of susceptibility, exposure and resilience (De Lange et al., 2009; Turner et al., 2003). This potential is determined by the ecosystem's characteristics that include different hierarchical levels, i.e., organisms, populations, communities and ecosystems (De Lange et al., 2010). Vulnerability finally encompasses intrinsic characteristics of an asset that render it sensible to degrade or potentially be lost (Metzger et al., 2008). These assets are valuable services or items that a specific society wants to protect because of the perceived benefits it obtains from them and could be lost (Gleyze, 2002).

In this study, the North-eastern Ecuadorian Amazon (NEA) is apprehended as a study case for evaluating the ecological vulnerability associated to its renowned status as a biodiversity hotspot even among intra-comparisons within the Amazon basin (Bass et al., 2010; Myers et al., 2000). For an illustration purpose, at least 210 mammals, 131 amphibians, 558 birds and 3,213 vascular plant total species richness, have been reported only in Yasuni National Park (Bass et al., 2010; Finer et al., 2008). Biodiversity is also potentially important for the development of medicines (Chaudhary et al., 2015; Neergheen-Bhujun et al., 2017). In addition, the NEA is of natural heritage importance due to its biodiversity, the continuing of ecological and hydrological functioning processes, and the integrity and relative uniqueness of species (IUCN, 2016). These territories are commonly monitored and managed; by legally effective means that intent to achieve their long-term conservation, and sustain their associated ecosystem services and cultural values (Dudley, 2008). Natural heritage is threatened by inappropriate anthropic development, when development-conservation conflicts arise, as established by the National Research Council (US) Panel on Biodiversity Research Priorities (PBRP, 1992).

However, in the NEA, a territory with such vast biodiversity, it is difficult to establish differences in ecological vulnerability when only non-exhaustive, spatially heterogeneous, biodiversity inventories have been fulfilled (Lessmann et al., 2016; PUCE, 2018). To evaluate this potential loss in case of hazard, the territory should first be evaluated homogenously (C. Mena et al., 2006), and a value might be given to each spatial unit, i.e., pixel or grid cell, therefore this value should indicate the potential loss or the relative vulnerability of a spatial unit over the others. The potential loss of ecological integrity and biodiversity, referred as total natural heritage, is abbreviated as PEB. The PEB is evaluated considering two spatial indices. Score values are given to each spatial unit (Marignani et al., 2017) based on conservation efforts represented by the designation of protection status (PS) and on potential biodiversity that is found for each land use category (BD).

In the NEA, ecosystem harmful impacts are propelled by demographic factors, (i.e., population growth and density), accessibility, (i.e. road density) and socio-economic drivers (i.e., oil industry, agriculture, forest logging, mining, inappropriate tourism, etc.) (Mena, 2008; Province GAD Orellana, 2011). These impacts are also exacerbated by shifting or absent adequate policies that increase pressure on land use, i.e., land tenure conflicts that result in protected areas boundary change, or in individually private or communal untitled lands (Holland et al., 2014; Mena et al., 2006; Sovacool and Scarpaci, 2016; Vallejo et al., 2015). Although, the forest cover studied remains more or less stable, with a deforestation rate reduced from 12% to 3% in the 2000-2014 time period (Holland et al., 2014), long-term nature conservation remains threatened by potential non-renewable extractive projects (Bonilla-Bedoya et al., 2014). Previous studies evaluated shifts in land tenure using protection area coverage as a proxy for assessing biodiversity vulnerability in space and time (Olson et al., 2001).

Biodiversity value has been previously successfully estimated from using the land use and land use cover/change proxy as it differs from one land use to another and from one land cover to another (Bernard and Fenton, 2002; Beukema and Van Noordwijk, 2004; Chaudhary et al., 2015; Haro-Carrión et al., 2009; Harvey et al., 2016, 2006; Lawton et al., 1998; Martínez et al., 2009). In particular, these studies indicated that species diversity generally declined with increasing anthropogenic alteration (Lawton et al., 1998; Martínez et al., 2009). Planning to maintain biodiversity and other natural values is essential and inherently spatial (Pressey et al., 2007).

Present research aims to (i) determine the degree of vulnerability, based on the level of ecological integrity, supposedly related to the more or less efficient protective measures within natural sites, and on the biodiversity level within land use classes, (ii) to map vulnerability, based on the scores obtained according to potential biodiversity and ecological integrity, and (iii) evaluate spatial congruence and correlation between putative ecological integrity and biodiversity. The main methodologies are scoring and spatial index methods, supported by overlay and correlation analyses. By building such vulnerability estimations at a regional scale, this study hopes to provide insights for spatial planning to improve adaptive capacity and governance of the natural heritage.

III.2. Materials and Methods

III.2.1. Study area

The study area (Figure 16) was restricted to the provinces of Sucumbíos and Orellana in the NEA (~144-900 m.a.s.l., Amazon lowlands), representing a 35,051 km^2 area (Figure 16B). This area includes upstream and midstream oil and gas production infrastructures which induce potentially polluting activities, likely to impact forests and aquatic ecosystems but also populations living in cities, villages and farms. This study excluded rivers with high flow rates (i.e., Napo, Tiputini, Coca, Payamino, Putumayo, Cuyabeno and Aguarico) where the surface waters and their interactions with groundwater are highly dynamic, therefore difficult or impossible to evaluate using scarce data. The study area is characterized by a warm climate with a temperature range of 20°C to 30°C, and an average annual rainfall of 2,900 $mm.yr^{-1}$ (Institute of Meteorology and Hydrology-INHAMI). The hydrology regime is irregular with 1,000 to 5,000 $m^3.s^{-1}$ daily discharges, and characterized by flash floods, due to high sensitivity to rain events (Laraque et al., 2007). The Intertropical Convergence Zone (ITCZ) is responsible for complex atmospheric processes in the NEA, which is influenced by interchangeable wind direction throughout the year, i.e., north to southward direction from October to April and south to northward from May to September (Palermo and Parra, 2014).

The territory is also characterized by eight indigenous populations from which two remain in voluntary isolation and have long time populated these areas. In addition, the studied area is rich in crude oil reserves, which have propelled the rapid colonization of these areas. The rich natural resources have prompted different chronological land use actions, resulting in land use changes and deforestation of the comprised ecosystems. Previous studies have indicated rapid deforestation rates from the 1980’s until the first decade in the 2000’s (Mena, 2008; Sierra, 2000) with decrease of the rate in 2014, trending towards stability (Bonilla-Bedoya et al., 2014).

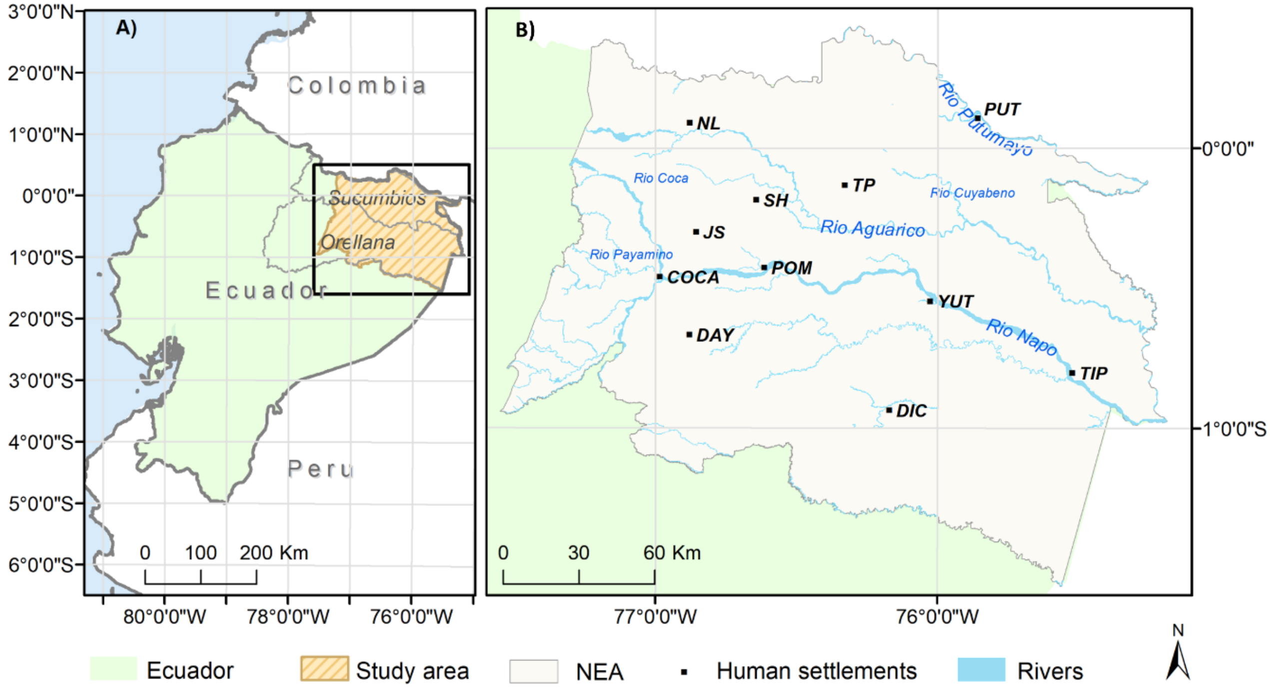


Figure 16. Location of the study area with rivers and human settlements. (A) The study area within the two provinces of Sucumbíos and Orellana in Ecuador, bordering Colombia in the North and Peru in the South. (B) The North-eastern Ecuadorian Amazon (NEA) including major rivers. Represented human settlements are abbreviated: DAY= Dayuma; NL = Nueva Loja (aka Lago Agrio); SH = Shushufindi; TP = Tarapoa; PAC = Pacayacu; POM = Pompeya; PUT = Putumayo; COCA = Puerto Francisco de Orellana; JS = Joya de Los Sachas; YUT = Yuturi; DI = Dícaro; SIN = Singue; TP = Tiputini.

III.2.2. Data compilation

The present research used Geographical Information System (GIS) layers about the natural heritage from Ecuadorian institutions such as Ministry of the Environment database known as the System of Environmental Information (SUIA for its abbreviation from Spanish), that are publicly available at http://mapainteractivo.ambiente.gob.ec/ , and the former Geographical Military Institute (IGM) which has, in recent years, merged with the Ecuadorian Spatial Institute, and compiled GIS databases for the entire Ecuador at different scales, available at http://www.geoportaligm.gob.ec/portal/. Table 13 summarises the spatial data compiled for this study.

Table 13. GIS database used to assess vulnerability of natural heritage in this study.

GIS data	Description	Sources
Land use	1 : 100,000	MAGAP-MAE-IEE (2014)
National System of Protected Areas	1 : 250,000 1 : 50,000	MAE (2014)
Socio Bosque Areas Forest Patrimony	1 : 100,000	MAE (2014)
Intangible zone Reserve of Biosphere	1 : 250,000	MAE (2008)

III.2.3. Vulnerability assessment

III.2.3.1. Assessing ecological integrity vulnerability

The vulnerability values can be assigned qualitatively, *i.e.*, user-, expert-, or researcher-defined (Ferraro et al., 2013). In the present study, the network of protected sites established within the NEA was used to evaluate the degree of ecological integrity, and thus the vulnerability of natural heritage. This approach is based on the two following assumptions:

(1) The level of protection granted to a given site is assumed to be a proxy of its ecological value/ecological integrity (both because the initial choice to protect the site was based on its ecological value, and because legal protection positively influences ecological integrity through time).

(2) Sites with a high ecological value are highly prone to natural heritage loss, and therefore more vulnerable in case of anthropogenic alteration (deliberate or not, legal or not).

The protection status (PS) levels were defined within the management plans in the study area, to which an ‘allowed’ level for anthropogenic interventions is conceived. These PS levels were defined according to relevant aspects, i.e., the level of biodiversity value, provision of ecosystem services and ecological functions that are intended to be protected in a given area (Ferraro et al., 2013). The protection level categories within the study area were defined via international regulatory institutions, United Nations Educational, Scientific and Cultural Organization (UNESCO, 2005), RAMSAR Convention (1975), in addition to few defined sites by

Local Autonomous Governments (GADs), which are detailed in Table B1. The PS was set to range from 1 (low) to 5 (high). See Table 14. A vulnerability degree was separately assigned to each site, however, if two or more PS overlapped, the value of the highest PS value was retained, because some pixels equally belong to biosphere reserve and national park or national park and intangible zones.

Table 14. Ecological integrity vulnerability index**.** Protection statuses of natural sites within the NEA with their corresponding level of anthropic development allowance and their given vulnerability value (see Figure 19 of the protection status in section III.3.2).

Protection statuses (PSs)	Allowed activities	Ecological Integrity Vulnerability Index (EIV)
National System of Protected Areas (SNAP), Intangible areas	None to limited human presence, maintaining of ecological and hydrological functioning processes, integrity and relative uniqueness; avoidance of any threat to their biodiversity (IUCN, 2016). Intangible Zone (ZITT) exceptionally exploitable if natural resources are considered of public interest (Finer et al., 2008; Vallejo et al., 2015).	5
Protected forests	Private, communal or public owned forests, with recreational, touristic, and scientific research purposes that are recognize by local decentralized governments (GAD) but not the SNAP.	4
Biosphere reserves, Forest patrimony	*Patrimony forests* are restricted land category with limited extractive activities such as farming or timber extraction, it cannot be private property or open sold to the market (Mena et al., 2006). They serve as buffer areas to larger reserves. *Biosphere reserves* are sustainable areas where human activities are allowed under sustainable practices. (UNESCO, 2017).	3
Communal Socio-Bosque	*The Socio Bosque Program* (PSB) was created to provide economic incentives to forested lands for at least 20 years (http://sociobosque.ambiente.gob.ec/node/174).	2
Zones without protecting status	All other land use areas dedicated to extensive agriculture, cattle ranching, urbanization, extractive activities, etc.	1

III.2.3.2. Assessing biodiversity vulnerability

III.2.3.2.1. Calculating biodiversity value from land use types

A land use map was built for Ecuador in 2014, based on combined multispectral satellite imagery: medium spatial resolution Land-Sat 8 and high spatial resolution Rapid-Eye, and ground-truth verification (MAGAP-SIGTIERRAS and Tracasa-nipsa, 2015). Various individual classes were defined by the Ecuadorian governmental institutions (the Ministry of Environment (MAE), the Ministry of Agriculture, the Livestock and Fisheries (MAGAP) and the Ecuadorian Spatial Institute (IEE) together known as the MAE-IEE-MAGAP coalition (IPCC Software, 2013; MAE and MAGAP, 2014)), in addition to the operational definitions given by the International Governmental Panel for Climate Change (IPCC). To facilitate vulnerability assessment, land use classes (LUs) were reclassified into 10 classes to match the LUs found in the literature with similar classes in the study area (see the procedure for biodiversity assessment in section III.2.3.2.2). Except for infrastructure and urban classes, which were regrouped into "human settlements", the other classes were kept as originally designated (Table 15).

Table 15. Definition of land use classes in the NEA.

Land use class	Definition (IPCC Software, 2013; MAE and MAGAP, 2014)
Primary forest	Plant community characterized by the dominance of trees of different native species, varied sizes and ages, with one or more strata.
Secondary forest	Regenerate on native forests, which have been cleared by natural or man-made causes, such as agriculture or ranching. They display a major difference in forest structure and/or species composition with respect to primary forests. Secondary vegetation is generally unstable, and represents intermediate successional stages.
Scrubland	Areas with a substantial component of native non-arboreal woody species. It includes areas degraded in transition to a dense canopy cover.
Grassland	Native herbaceous vegetation with spontaneous growth, used to sporadic cattle ranging, wildlife and conservation, which do not need human managing.
Agricultural mosaic	Group of cultivated species that are mixed altogether and cannot be individualized; and exceptionally may be associated with natural vegetation.
Perennial crops	Crop lands with a vegetative cycle of more than 3 years that may be harvested several times a year.
Semi-perennial crops	Crop lands with a vegetative cycle between 1 and 3 years.
Pasture	Herbaceous vegetation dominated by *Poaceae* and leguminous introduced species, used with cattle ends, which for their establishment and conservation need human managing.
Annual culture	Crop lands with seasonal vegetative cycle, that may be harvested several times a year
Human settlements	Built environment encompassed by urbanized areas, artificially made with high human population density and related infrastructure.

To calculate the potential vulnerability to biodiversity loss for each LU, it was necessary to provide a current biodiversity value (BDv) associated to them first. This value associated to land use is named the biodiversity vulnerability index (BVI). A higher BDv means that greater biodiversity is found per unit of surface in a given LU. The method for assigning BDv to LU was based on the species richness of vascular plants (Marignani et al., 2017; Schmidt, 2008). Derived BDv from Neo-tropical LUs were obtained from a combined meta-analysis and expert surveys previously carried out by Henzen (2008).

As for the ecological integrity vulnerability, the assumed hypothesis is that the BDv of a LU is positively related to the risk of biodiversity loss in case of anthropogenic alteration. Higher vulnerability values were thus attributed to the more biodiverse LUs.

Each LU obtained a value from 0 (low) to 5 (high), calculated via Equation 7 (Henzen, 2008):

$$BDv = (5.Sh^{-1}) \times Ss, \qquad \text{(Eq.7)}$$

Where:

BDv = the biodiversity value (0-5) of a sample, determined for each land use category.

Ss = the number of species observed per hectare for a given land use category

Sh = the highest species number per hectare among the different categories of land use.

III.2.3.2.2. Validity of the relationship between biodiversity and land use

The representativeness of vascular plant species as a proxy for total species diversity in the neo-tropical region was tested by comparing the BDv obtained from vascular plant species richness in the neo-tropics (using the meta-analysis and expert opinion described in section III.2.4.1) to multiple and cross-taxa species richness (taxonomic groups studied; n = 27) in similar land uses across pan-tropical regions, because there were not enough available studies in the neotropics. The neotropics in this case refers to the American continent, encompassing zones within the tropical latitudes, exclusively to countries south from Costa Rica to Northern Peru. A total of 101 plots in different LUs were selected from a set of 40 publications and a biodiversity value was calculated to each of them, following the procedure described in section III.2.4.1. In order to do so, a non-parametric Kruskal-Wallis test, followed with *post-hoc* Wilcoxon-Mann-Whitney tests with Bonferroni corrections, were used to compare species richness among LUs, because of evident heteroscedasticity and departures from normality, precluding the commonly used analysis of variance (ANOVA) (Zar, 1974). Following, a Spearman's rank correlation was performed (Spearman, 1907), in order to determine the relationship between the ranks given to each LU according to the diversity of either vascular plants in the neo-tropical zone or other taxa in the whole pan-tropical zone, including African and Asian countries. All analyses were performed using R software (R Development Core Team, 2008).

III.2.4. Standardisation and combination of vulnerability values

A spatial analysis was conducted to ensure standardization and comparability of measurements from the two resulting maps, *i.e.* vulnerabilities based on ecological integrity or biodiversity value. Biodiversity values obtained were transferred into ArcGIS® system. The biodiversity value map was rescaled from a continuous 0.01 to 1.00 scale to a qualitative scale (index scores) from 1 to 5 to match the ecological integrity vulnerability map.

Two spatial analyses were performed by combining the two standardized vulnerability maps, using simple overlay operations:

1) Considering the two vulnerability maps as independent parameters or attributes of the same system, the spatial addition of the biodiversity and ecological integrity indices, at each pixel, results in a more integrated vulnerability score. The spatial operation of adding vulnerability levels per pixel therefore results in values that range from 2 to 10. It may add up to as many integrated attributes as the researcher needs.
2) Spatial difference between the ecological integrity – biodiversity index maps may assess whether land use categories with high biodiversity values are within zones regarded as having a sufficient degree of protection and thus capable of maintaining their biodiversity in the long-term (Ferraro et al., 2013). The values can range from -4 to 4. Total spatial congruence between the two metrics will return null values. Negative values indicate situations where a high protection status is not associated to high biodiversity, while positive values indicate situations where high biodiversity is not well protected. Per unit decrease or increase reflects the intensity of spatial incongruence.

III.3. Results

III.3.1. Biodiversity vulnerability map obtained from land use

III.3.1.1. Species richness across land use categories in tropical regions

The biodiversity vulnerability index (BVI) was calculated using neotropic vascular plants as surrogate. The Kruskall-wallis test showed significant differences in terms of species richness among LUs (χ^2 = 31.3, df = 3, $P < 0.0001$). The Wilcoxon-Mann-Whitney test indicated that primary forests (with 80 vascular species per hectare on average) differ significantly from other land uses ($P = 0.0008$). Score ranks obtained for the different land use categories with data from either vascular plants in neotropic or multi-taxa in pan-tropical regions were exactly the same (ρ = 1.00, n = 4) but, due to the low number of items used for this calculation, the correlation was only marginally significant ($P = 0.0833$). The data analysis for the BDv comparison across taxa in order to validate the BVI showed differences for primary forest (only 0.27 lower score for multiple-taxa compared to vascular plant data). The BVI for other land use classes showed lower BDv for vascular plants in the neotropics compared to pan-tropical multi-taxa (0.96 - 1.00 score variation). In general, the ranking of BDv categories remained unchanged across land uses. Table 16 compares average BDv obtained from multi-taxa or neo-tropical vascular plant species richness among land use types.

Table 16. Biodiversity values of four land use (LU) types calculated from the species richness of vascular plants in the neotropics (used in the present study to derive Biodiversity Vulnerability Index) or of multiple taxa in the pantropical regions (based on a literature review of 8-40 studies depending on the LU).

Biodiversity Value	Primary forests	Secondary and logged forests	Perennial cultures	Pastures
Vascular plants in neotropic	5	3.2	2.26	1.3
Average calculated across multiple taxa	4.73*	4.14	3.32	2.3

(n = number of studies)	(n =40)	(n=27)	(n=26)	(n=8)

*Significance at 0.01%

III.3.1.2. Relative vulnerability values across the landscape

The land use map in 2015 (Figure 17) indicates that primary forests account for the potentially 'highest' vulnerable area, followed by secondary forests with 'medium high' vulnerabilities. Surface extent of vulnerable areas were from the highest to the lowest: 70.1% (14 390 km²) high, 12.4% medium high, 6.7% medium, 8.2% medium low and 0.3% low (Figure 18). Table 17 shows the biodiversity value (BDv) in the neo-tropical region for the ten LU categories.

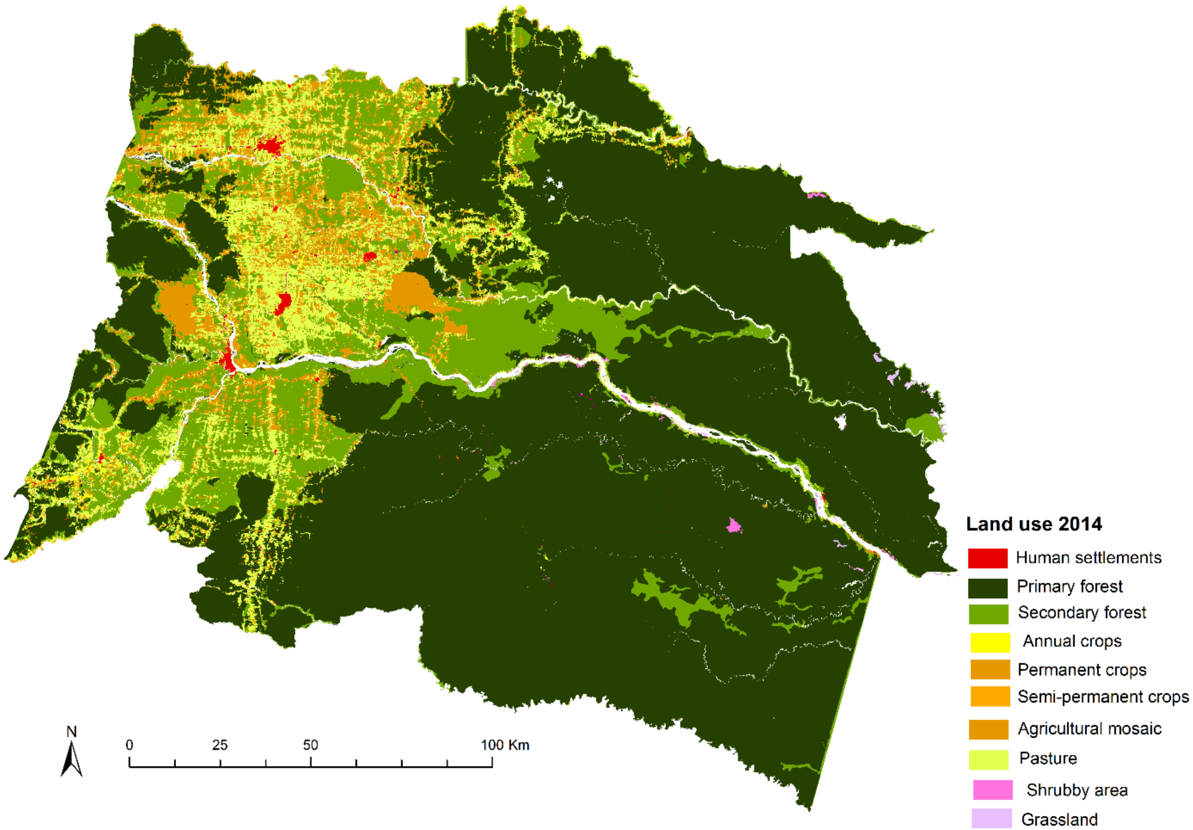


Figure 17. Classification of land use according to MAGAP-MAE-IEE (2014). Color hues selected from combination of the CORINE and the Inventaire National Forestier (INF) land cover systems (IGN, 2018) to ease distinction between shrubby lands, grasslands and annual crops.

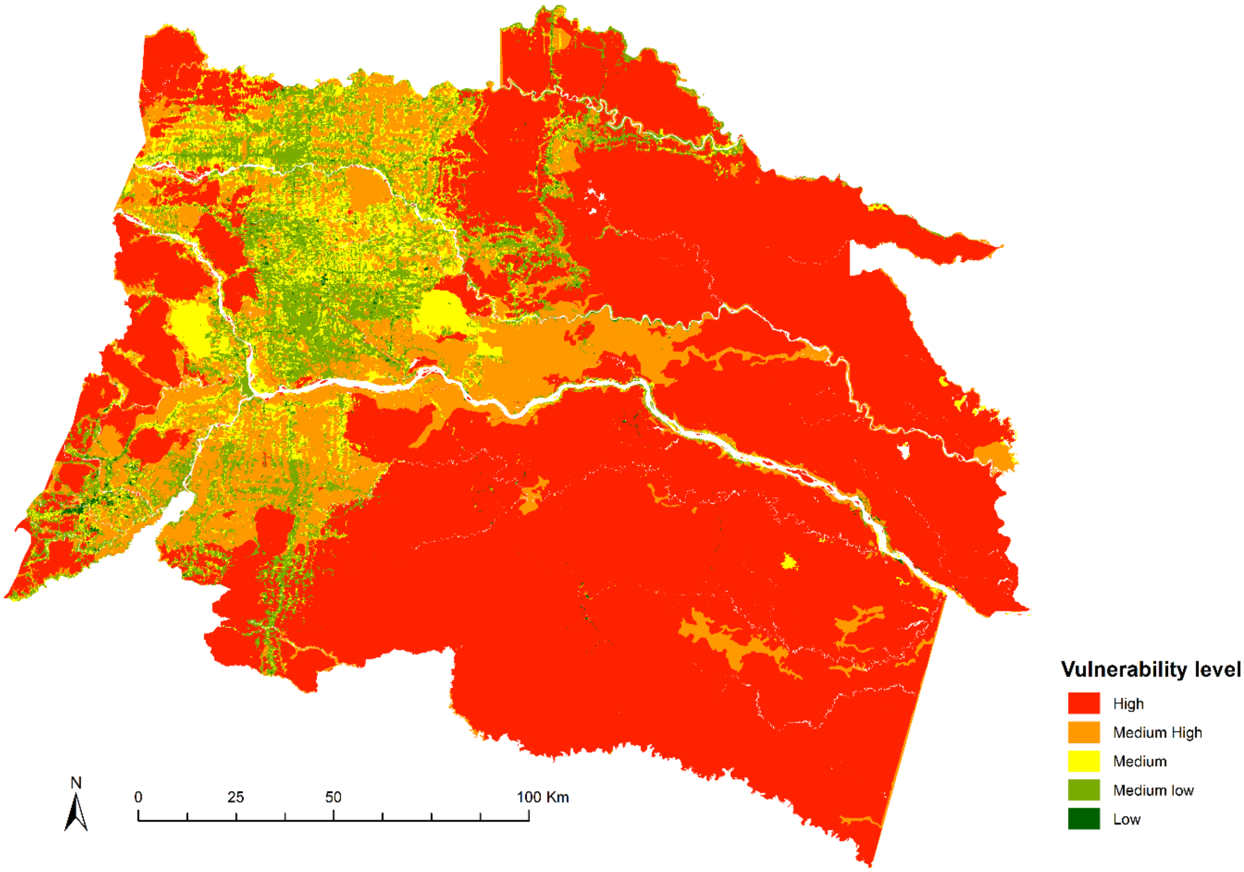


Figure 18. Ecological vulnerability according to potential biodiversity derived from land use in 2015.

Table 17. Land use classes with their relative surface, biodiversity value and corresponding vulnerability level and Biodiversity Vulnerability Index (BVI).

	Biodiversity value calculated (BDv)	**Relative surface (%)**	**Vulnerability level**	**Biodiversity Vulnerability Index (BVI)**
Primary forest	5.00	70.07	High	5
Secondary forest	3.20	12.40	Medium High	4
Shrubby areas	2.53	0.12	Medium	3
Agricultural mosaic	2.40	3.02	Medium	3
Permanent crops	2.25	3.40	Medium	3
Semi-permanent crops	2.25	0.05	Medium	3
Grassland	2.08	0.08	Medium	3
Pasture	1.30	8.16	Medium low	2
Human settlements	1.13	0.43	Medium low	2
Annual crops	0.89	0.25	Low	1

III.3.2. Ecological integrity vulnerability map obtained from protection status

A vulnerability map based on putative ecological integrity was obtained from protection status layers. This was analyzed as a single merged spatial layer (Figure 19). The different vulnerability classes represented variable surfaces, in a descending vulnerability order: 41% (14,390 km^2) high, 6.6% (2,312 km^2) medium high, 21.4 % (7,495.8 km^2) medium, 0.3% (96 km^2) medium low, and 30.7% (10,757.2 km^2) low (Figure 20). This is notably linked to the geographical location and the size of the two national parks: the Yasuní National Park and the Cuyabeno Wildlife Reserve. Human density concentrates on the western and center zones, where no protected site had been declared except for the Biosphere Reserve of Sumaco.

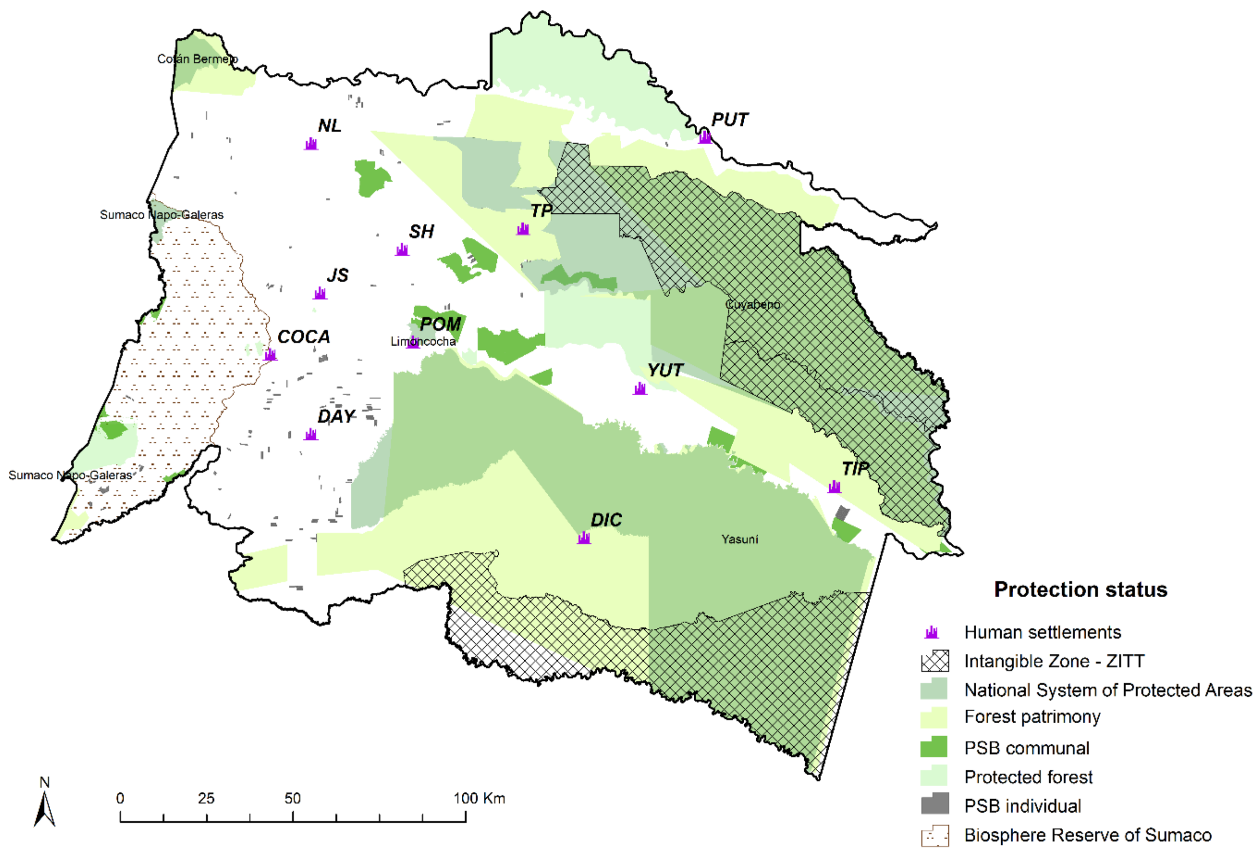


Figure 19. Distribution of protected areas in the NEA. White colored areas indicate no protection status.

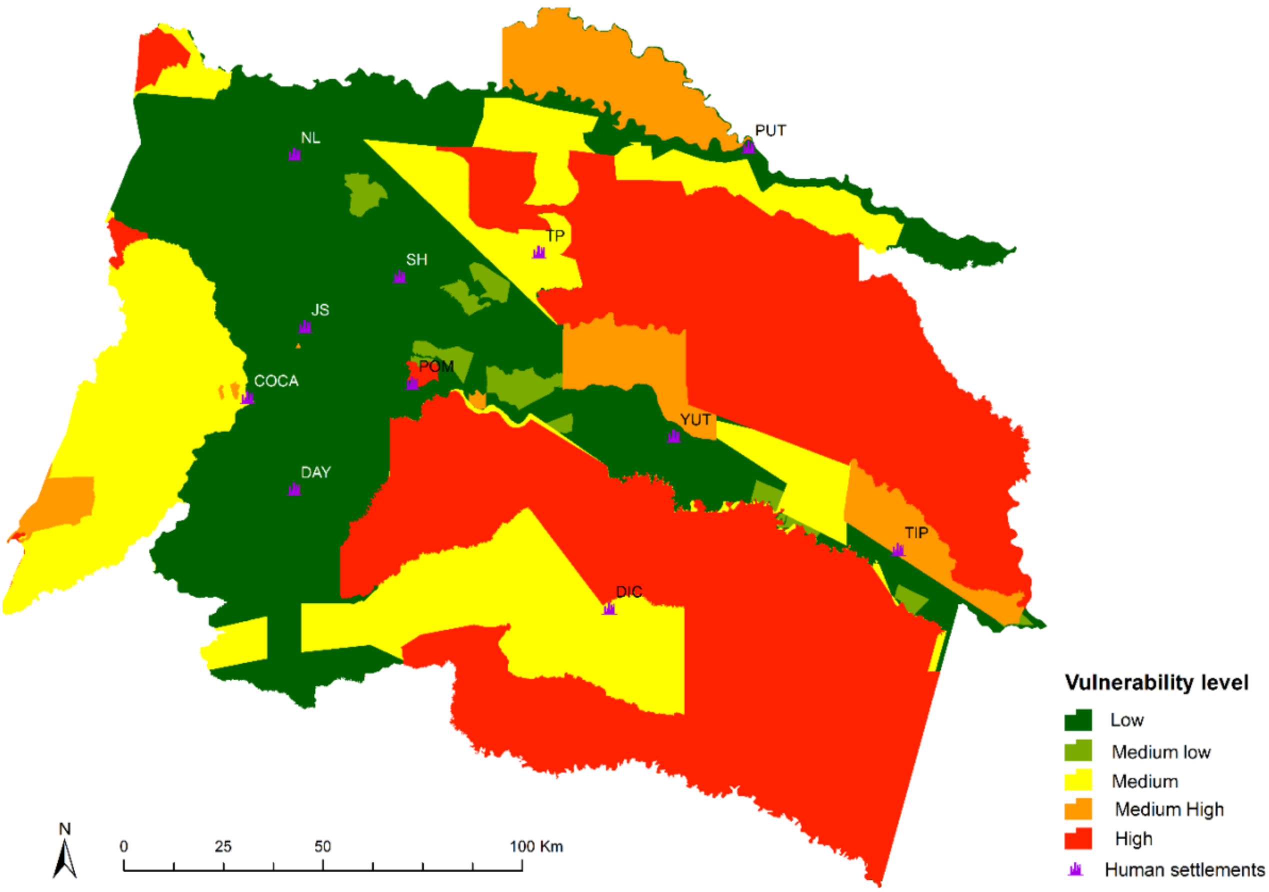


Figure 20. Vulnerability levels according to the legal status of nature protection.

Table 18. Surface extension of the different protection statuses shown with corresponding relative percentages in the NEA (35,051 km^2). The sum of values for all statuses does not make total due to a surface overlapping of 26.6%, as several sites are protected by more than one legal status.

Protected Area (PA)	**Surface (km^2)**	**Relative surface to the entire area (%)**
SNAP	12,876.2	36.7
Individual Socio-Bosque	117.8	0.3
Communal Socio-Bosque	1,749.3	5.0
Biosphere reserves	2,792.9	8.0
RAMSAR	46.1	0.1
Forest patrimony	16,818.1	47.0
Intangible zones	7,667.1	21.9
Protected forests	2,312.9	6.6
Non protected area	10,499.1	29.9
Total protected area	24,551.9	70.1
Total	35,051	100

III.3.3. Relationships between vulnerabilities derived from the different metrics.

The spatial correlation between the maps resulting from the two approaches considered (Figure 21) indicates globally a moderate positive correlation (r = 0.52). 3,217 km^2 of unprotected primary forest tended to decrease this correlation value. In contrast, significant surfaces appeared in yellow in Figure 21, corresponding to areas with equivalent vulnerability levels whatever the metric used. In general, 42% of the NEA was considered as highly congruent between potential biodiversity and protection status.

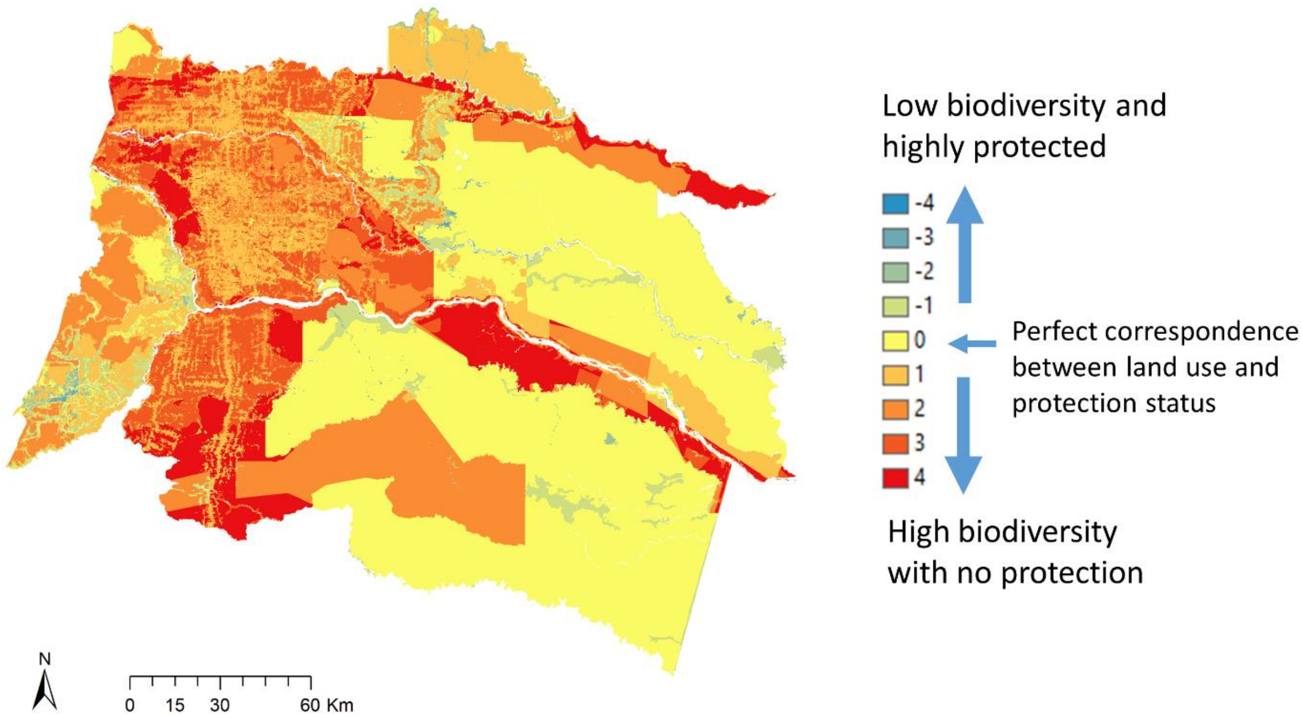


Figure 21. Difference between ecological integrity and biodiversity vulnerabilities. Positive values indicate high vulnerability according to potential biodiversity but with a low protection status. Negative values indicate low vulnerability according to potential biodiversity, but with a high protection status. Null values indicate an agreement between potential biodiversity and protection status.

Land use analysis indicates that less than one percent of annual crops and pastures, harboring the lowest biodiversity, was highly protected (in blue in Figure 21) in the south-western zone within the Sumaco Biosphere reserve where only sustainable socio-economic activities are allowed, and in eastern areas where prohibition of all human activity is expected within Yasuní and Cuyabeno reserves.

Table 19. Spatial congruence analysis of overlapped zones between Protection status and Land Use.

Overlapped zones (ecological integrity and land use)	Congruence value (difference)	Area (km^2)	Contribution (%)
No protection on PF	-4	3,217.3	8.8
No protection on SF	-3	3,430.7	9.4
SB, FP or BR on PF or SF	-2	7,658.7	21.7
BR, FP on AgM	-1	4,5023.5	13.9
YNP, ZITT or CBR on PF	0	14,690.7	42.0
SB on AgM	1	1,284.3	3.8
SB or ProFor on Pasture	2	242	0.5
ProFor on AC	3	58	0.1
YNP, CBR or ZITT on AC	4	4.8	0.02
Total overlapped zones	-	35,051	100

AC = Annual crop; AgM = Agricultural mosaic; BR = Biosphere Reserve; CBR = Cuyabeno Biological Reserve; FP = Forest Patrimony; EIV = Ecological Integrity; PF = Primary forest; ProFor = Protected forest; SF = Secondary forest; SB = Socio-Bosque; YNP = Yasuni National Park; ZITT = Intangible Zone.

The natural heritage vulnerability evaluated with the PEB would actually be the addition of both spatial indices. Thus, the highest PEB areas encompass the overlapping zones of the ZITT, Yasuní National Park and Cuyabeno Reserve with primary forest. The lowest values are non-protected areas with annual crops and human settlements zones. Medium vulnerability may correspond to multiple combinations of the BDv and SP, i.e., primary forest overlapping with non-restricted zones may have the same value than a Socio-Bosque communal area that overlaps secondary forest; these combinations are displayed in Figure 22.

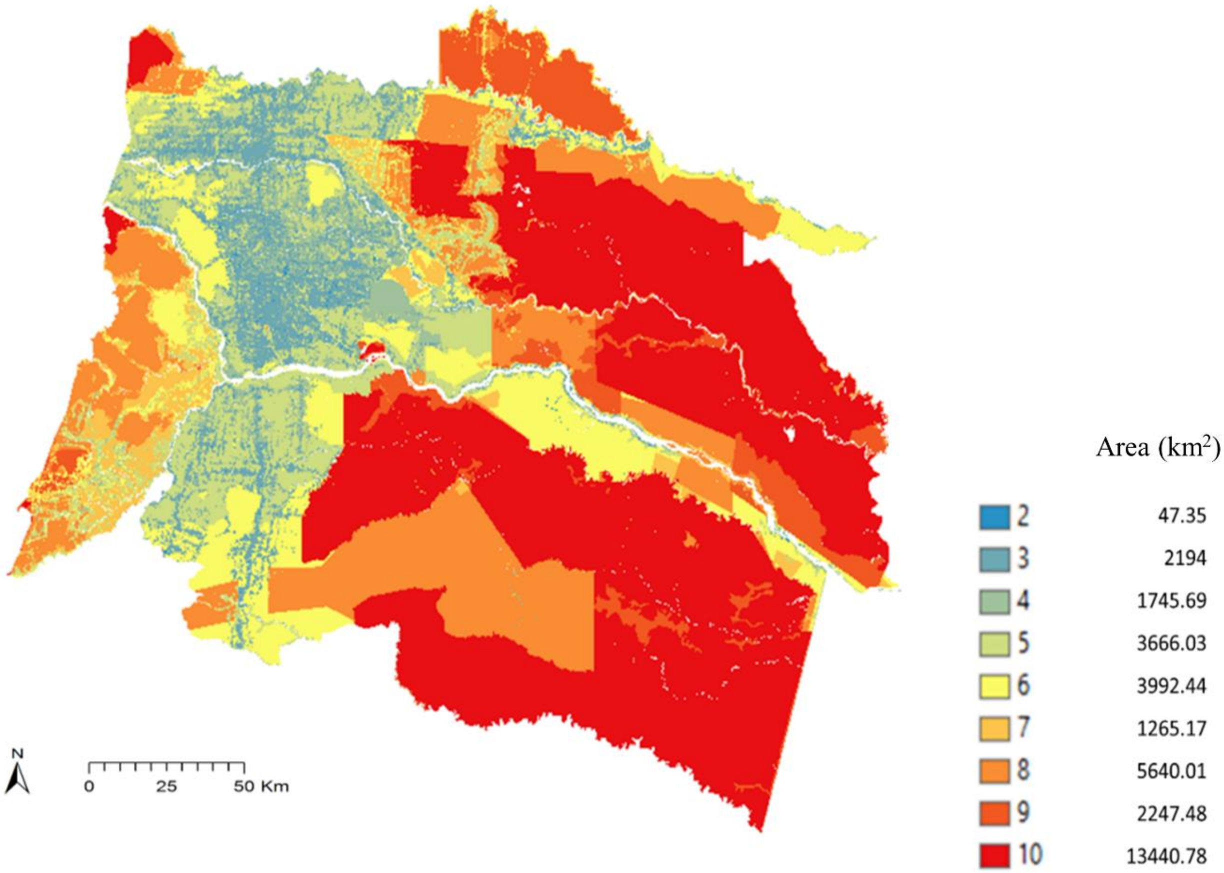


Figure 22. Addition of ecological integrity and biodiversity indices representing the overall vulnerability of natural heritage assets in the NEA (referred to in the text as *potential loss of ecological integrity and biodiversity*, abbreviated as PEB).

III.4. Discussion

III.4.1. Relevance of protection status in vulnerability assessment

The present vulnerability scoring system allows the mapping of vulnerability regarding natural heritage using spatial indices. Two spatial indices for biodiversity and ecological integrity, abbreviated PEB, were developed along a homogenized regional scale. The relevance to use protection status as a surrogate of ecological integrity is also because nature conservation can be inefficient even in protected areas; some authors use the expression of “paper parks” to describe protected sites prone to situations that undermine the actual conservation of nature within their jurisdiction (Bonham et al., 2008; Di Minin and Toivonen, 2015). However, previous studies conducted in the NEA with remote sensing techniques have indicated high levels of long-term conservation of the natural heritage when relating land use change to the protection status (Messina, 2006). This strengthens the hypothesis that protection status can be used as a proxy for ecological integrity. Hence, evaluating from protection status the potential loss of natural heritage upon plausible future hazards is pertinent to assess vulnerability (Adger, 2006). Such hazards could be the release of toxic compounds by oil industry or other anthropic activities. These are seldom occurring in protected areas, but (1) our aim was to assess intrinsic ecological vulnerability, regardless of potential hazards, and (2) one cannot take for granted that the human activities allowed in the protected areas will not change in the future, through the modification of their boundaries or because of a change in the perception of which economic activities are compatible with nature protection. Finally, the main limitation of this approach concerns on the contrary some unprotected areas yet occupied by primary forests. This is why it is important to associate this approach to complementary assessment based on land use.

III.4.2. Biodiversity and land use

The usefulness of the BVI depends on the capability to rank the land use classes by their biodiversity value, which in turn is given by the capability to measure actual biodiversity. First, the use of vascular species as proxy for species richness has been supported by previous studies (Estrada et al., 1993; Kessler et al., 2016). It was also in agreement with another study that considered multiple taxa and showed that species richness decreased proportionally to the degree of modification of habitats (Schulze et al., 2017); the validity procedure conducted in the present study also supported this statement. These studies have shown that species richness was higher in primary forests than in secondary forests or agricultural mosaics; moreover, primary forests are particularly important for some geographically restricted species (Dumbrell and Hill, 2005) and specialist species that can only thrive in narrow and restricted ecological niches and with a limited diet (Faria et al., 2006). The marginally significant differences in terms of species richness among medium vulnerability classes can be explained because biodiversity is associated to trees in secondary forest or agricultural mosaics. Many species might transiently use these zones as corridors (Eggleton et al., 2016; Gardner et al., 2007; Medellín and Equihua, 1998). Conversely, some studies have also shown that species richness is not necessarily higher for primary forest compared to secondary forests (Faria et al., 2007; Hawes et al., 2009; Martínez et al., 2009), yet, proximity to primary forests probably plays an important role (Hawes et al., 2009). Agricultural mosaics are higher in biodiversity than annual crops probably because of mixed crops and woody plants on borders (Holland and Fahrig, 2000).

III.4.3. Implications for spatial planning and conservation policies

Land use management and planning should integrate PEB indexes for accounting vulnerability of natural heritage. The PEB index derived for ecological vulnerability assessment may also help managing the protection of some environmental assets such as ecosystem services via long term ecological integrity (Huq et al., 2019). Similarly, the spatial difference tool indicates zones where protection status level is incongruent with biodiversity value scores (i.e., some low scored annual crops are highly protected, while some highly biodiverse primary forests are not protected). In summary, this analysis indicates at least three main areas for spatial planning. The conservation priority points out to primary forests, yet old secondary forests and agricultural mosaics are seen as having also an important role for conservation (Schulze et al., 2017), notably because fragmented forested areas are known to deprive fauna and flora genetic pools (Foster et al., 2013; Silbert, 2002). Specifically, connectivity management may attempt to reconnect habitats from east to west areas in the NEA whose PEB values and scores are highest. In addition, related development infrastructures and socio-economic activities are drivers of deforestation. In the NEA, the oil industry is responsible for the most relevant sources of pollutant discharges to the environment (MAE-PRAS, 2016). In this sense, conservation priorities should consider the links of ecosystems within these land covers, and related ecosystems because they can influence local biodiversity (Cebrián-Piqueras et al., 2017).

Oil activities derived from governmental policies have been the main underlying cause of anthropic interventions to undisturbed areas. But the main proximate causes of deforestation at the parish level are road and population densities (C. Mena et al., 2006), type of road access (Baynard et al., 2013) and land tenure conflicts (Holland et al., 2014; Messina et al., 2006). The National Secretariat for Planning and Development (SENPLADES) developed a spatial atlas with socio-economic indicators which indicates that social development is heterogeneous in the NEA region with impoverished areas being potentially overlapped with ecologically vulnerable areas, increasing the risk to loss for the natural heritage, as natural resources are coveted income sources (SENPLADES, 2013).

III.4.4. Caveats of the study and steps forward

Regarding the BVI, other metrics might be used to sensing biodiversity values according to land uses. An alternative may be to use species diversity, species evenness or assemblage composition (Martínez et al., 2009). Other possibilities could be proposed, such as mapping biodiversity variability according to local perceptions, or through pixel neighborhood impacts. Neighborhood impacts in a pixel can be evaluated using weighted averages of border effects, fragmentation or proximity to other land uses that influence biodiversity variability (Brinck et al., 2017). Assessment of biodiversity requires further research on more taxa to be able to improve the capacity to rank land uses and vulnerable categories. Biodiversity conservation priorities can be assessed using ecological knowledge or social and political values (Egoh et al., 2011). However, there is an indirect spatial overlap with human well-being, via the notion of ecosystem services (Maes et al., 2016). Contrasting results have been reported using spatial overlay methods, i.e., positive low correlations or no correlation have been reported between biodiversity and ecosystem services (Egoh et al., 2009). Furthermore, a time series analysis of land use and cover change, combined with the evolution of protection statuses, may help to describe the changes in past and future potential losses using chronological shifts in protected site boundaries and land use changes in order to calibrate the PEB index.

Another important aspect of vulnerability is resilience. Resilience indicates the capacity of response of a system to regain a functional state after a disturbance (Becerra et al., 2016; Walker et al., 2002). The ecological resilience in the NEA can be addressed with the capacity of the system to restore net primary production (Vogt, 2015). System alternate states are evaluated via thresholds (Resilience Alliance and Santa Fe Institute, 2018), thus they should be integrated in order to complete the evaluation of vulnerability. Vogt et al., (2016), have already defined resilience thresholds in Brazilian Amazon forests by combining precipitation average values and soil types. These evaluations can serve as a progress towards more complete vulnerability assessments. Despite the many hindrances from available databases (Resilience Alliance and Santa Fe Institute, 2018), sufficient data are available in the NEA to investigate resilience from the angle of net primary production.

III.5. Conclusions

Several studies have depicted the socio-ecological systems, nature conservation discourses, and deforestation frontier settings characteristics of the NEA (C. Mena et al., 2006; Messina and Walsh, 2005; Walsh et al., 2008a). These studies have emphasized the negative impacts of deforestation in natural heritage driven by potential anthropic interventions. In present research this information in included to assign vulnerability indexes to diverse spatial units within the NEA by integrating homogeneous regional spatial data to develop a vulnerability index. The result is the PEB index, to diagnose and identify areas with insufficient protection, mainly those at higher loss potential due to human interventions. Furthermore, a time series analysis of land use and cover change vs. status of protection might help to describe the causal-consequence relationships between protection boundary changes through time and space with land use changes to predict and reinforce the hypothesizes established in this study. Other complementary parameters could be integrated within environmental vulnerability assessment, i.e. potential drinkable water resource. In a next step, the PEB and key hazards (Marignani et al., 2017), i.e. mining and crude oil extraction (Baynard et al., 2013) or population growth/agriculture expansion (C. Mena et al., 2006) may be combined for risk assessment (Birkmann, 2011; Gleyze, 2002; Kazakis and Voudouris, 2015).

From vulnerability of biodiversity and natural heritage to the risk of groundwater to oil pits

In chapter III, the implemented proxy-based method permitted, to an acceptable degree, to assess natural heritage vulnerability. Using two proxies in the form of maps at regional scales aimed to assess ecological integrity and biodiversity, considered as important assets in the NEA. Spatial overlay revealed some insights for land use planning. Thereby, detecting the relevant allocation of protection measures to zones differing in potential biodiversity is important for an efficient protection of natural heritage and for future land reallocation to other uses. Furthermore, the PEB, an overlay-index map, was developed using the available spatial and homogenous information of the public data from Ecuadorian institutions.

Many other environmental assets can be included in a vulnerability assessment or risk analysis. It was perceived that spatial land planning in the NEA regarding its groundwater vulnerability may be important to analyze, because hazards from unlined oil pits may threaten groundwater that could be potentially contaminated by oil-associated pollutants. Chapter IV deals with the third step of an ERA (environmental risk assessment) that results from the combination of both vulnerability maps and hazard maps. An ERA can be evaluated for single or cumulative hazards and many combinations are possible, depending also on the vulnerabilities considered.

One example is used in this study, using a remaining hazard source: unlined oil pits, which seems to be of greater pollution potential risk to groundwater than the other two hazard sources already evaluated, due to its proximity to groundwater. In the NEA, oil industries refuse to acknowledge for past waste stores and unintentional discharges from unlined oil pits (Buccina et al., 2013), which are considered as major sources of risk of contamination for hydrogeological systems (Hurtig and San Sebastián, 2002; Wernersson, 2004). Risk of groundwater contamination is a major concern for local population, who allege oil-related contamination to cause illnesses, resulting in unprecedented international litigations (Buccina et al., 2013; Kimerling, 1990).

Chapter IV - Risk assessment of unlined oil pits to groundwater quality in the Ecuadorian Amazon: A modified GIS-DRASTIC approach

Unlined oil pit Shushufindi #61 dubbed "the Presidential pit"

"Not taking risks one doesn't understand is often the best form of risk management."

—**Raghuram G. Rajan**

Article in preparation

Highlights

- Aquifer vulnerability was assessed with modified DRASTIC model in forested and crop lands.
- Oil pit hydrocarbon contents and volumetric dimensions were inputs for hazard analysis.
- Risk was derived from hazard and vulnerability using overlaying and cost distance analyses.
- Robustness of parameters was validated by sensitivity analysis.
- Finer scaled research is suggested in highest risk zones shown by this study.

Abstract

Oil waste stored in unlined oil field waste pits (OP) are considered as potential source of contamination to the amazon hydrological systems and landscapes. Risk of groundwater contamination in the North-eastern Ecuadorian Amazon (NEA) may be important due to seepages from OPs. Previous studies allege contamination related to oil production has caused illnesses, triggering unprecedented international lawsuits. Risk can be spatially defined by the overlaying of vulnerability and hazard. Two maps encompassing forested and agriculture mixed lands were, hence, produced using available unified/homogenized geographical datasets from various sources: (1) a vulnerability indexed map using a modified DRASTIC model, associated to sensitivity analysis to evaluate the variation of its parameters; (2) a hazard map, based on Total Petroleum Hydrocarbons (TPH) contained in oil pits, associated to a cost distance analysis for different risk maximum distance ranges (MDRs) to model the contamination plume. A median TPH concentration was calculated from available data and used to derive total volumes of TPH within OPs, using soil structure types, oil densities and OP volumetric dimensions. Results indicate that aquifers at highest risk are located in Sucumbíos and Orellana provinces. Two models' parameters, final recharge and depth to water, have highest weight (W=5) and appear to be the most sensitive (mean variation index = 1.9% and 2.1% respectively). The amount of TPH calculated per pit (90.5 kg on average) was highly variable across oil pits (min = 5.1 kg; max=3.3 t). The median TPH concentration of 7,519 mg.kg^{-1} was used for extrapolation to oil pits with no available data. Risk using MDRs from 0.5 to 5 km were mapped. Spatial risk assessment drew insights for improved environmental monitoring and public health associated land use planning.

Keywords: risk assessment, overlay-index, Amazon, groundwater, oil extraction, land use planning, hydrocarbons

IV.1. Introduction

In the North-eastern Ecuadorian Amazon (NEA) oil productive activities are alleged to have released hydrocarbons to the ecosystems, potentially causing human health issues (San Sebastián and Hurtig, 2005). Studies on contamination by total petroleum hydrocarbons (TPH) indicate high concentrations in soils, sediments, drinking water and surface water, that communities have been using for daily life activities and recreational purposes (Barraza et al., 2018; Maddela et al., 2016; Merchán-Rivera et al., 2017; Wasserstrom, 2013). A comparative study on water environmental compartments, indicated groundwater to be the highest contaminated amongst all (Merchán-Rivera et al., 2017) and suggested to be linked to cancer mortality (San Sebastián et al., 2001a).

A relevant contaminant source is related to historical liabilities such as: unlined earthen reserve, mud drilling and waste oil pits (hereafter named pits). A study in Ecuador claimed that until 1993, oil producers, from which Texaco was by far the most prominent , while extracting oil production, built and later abandoned between 800 and 1000 pits (Buccina et al., 2013) where 505,600 t of crude oil have been poured (MAE-PRAS, 2016). During the first Remedial Action Plan (RAP) in 1995, only 158 pits have been merely covered with a barren soil coating (MAE-PRAS, 2016).

The purpose of a pit is to hold drilling cuts, drilling mud and crude hydrocarbons that are not collected by the pipeline (MAE-PRAS, 2016). Their construction standards require particular volumetric dimensions, i.e., size should be sufficient to ensure adequate storage until closure, taking into account historical rainfall patterns and the fact that depth should be such that the bottom does not contact the groundwater table, so the pit contents do not adversely impact groundwater or surface water. A site chosen for implantation of pits should be investigated in terms of presence of aquifers and a specific action plan should be elaborated to protect them (EPA, 2014). The Programme of Environmental and Social Remediation (PRAS) initiated in 2018, registered more than 2000 pits until 2013 from past (from which between 800 and 1,000 belonged to Texaco) and present oil production operations with, for the 1420 pits documented, high variability in volumes (see section III.2.5).

The potential contamination of groundwater should be investigated by identifying vulnerable zones across the NEA, i.e. zones which would be more impacted by occasional or chronicle contamination from oil pits. Vulnerability refers to the degree to which a system, subsystem, or system component is likely to experience harm due to exposure to a hazard or environmental change, either from a perturbation or a stressor (Turner et al., 2003) and can be classified as intrinsic and specific vulnerability (Natural Research Council, 1993). Intrinsic vulnerability of an aquifer is defined as its susceptibility to be reached and affected by pollutants from the surface that diffuse into the groundwater (Vrba and Zaporozec, 1994). Specific vulnerability can be defined as the susceptibility of contamination of the aquifer by a particular contaminant, incorporating its own physicochemical properties (Andreo et al., 2006). Then, risk to groundwater can be spatially defined by the probability of contaminant leaching or hazard, overlaid with vulnerability (Kazakis and Voudouris, 2015; Metzger et al., 2008). It is relevant to quantify hazard from contaminant concentrations by overlaying with vulnerabilities, i.e., other studies restrain only to model potential hazard from land use (Kazakis and Voudouris, 2015; Mimi and Assi, 2009).

There are several spatial explicit models to assess groundwater vulnerability. Aller et al., (1987) developed the DRASTIC model (*i.e.* D=depth to water; R=net recharge; A= aquifer media; S= soil media; T= topography; I= impact of vadose zone; C= hydraulic conductivity) which seems to be more comprehensive, as it assesses groundwater vulnerability, indexing more information regarding contamination potential to reach groundwater, than other GIS based methods such as SI GOD, AVI and SINTACS (Gogu and Dassargues, 2000). The information is organized into seven parameters assessing intrinsic groundwater vulnerability with rating scores and weights (Aller et al., 1987). The DRASTIC model has been further modified since its development, depending on available spatial data layers (Al-Adamat et al., 2003; Merchant, 1994; Mimi and Assi, 2009; Stigter et al., 2006). In the Daule aquifer, in Ecuador, groundwater vulnerability was assessed using a reduced set of parameters of the DRASTIC index model due to several shortcomings, intrinsic to the model (Merchant, 1994) or site specific (Ribeiro et al., 2017). Some studies have incorporated land use types (Mimi et al., 2012), including industrial related contamination (*e.g.* oil and gas exploitation), turning the model from intrinsic to specific vulnerability (Kazakis and Voudouris, 2015; Mimi and Assi, 2009; Secunda et al., 1998) which integrates specific physicochemical characteristics of the pollutant. Other studies have replaced some of the original parameters with others more specific to the context or to data availability, for instance, replacing hydraulic conductivity with groundwater velocity (Kazakis and Voudouris, 2015).

This study aims to (1) define vulnerability levels for aquifers throughout the NEA, (2) to estimate and map hydrocarbon content in pits and their associated hazard level, and (3) to evaluate the risk of pit leakage to groundwater along the runoff stream network with indexed vulnerability map overlaying hazard level map, using a regional contamination propagation plume. Several MDRs of distance dispersion were explored coupling TPH concentrations and volumetric properties of pits. This study aims to provide insights for improved land use planning, public health recommendations and future detailed site specific investigations in this historic oil producing region.

IV.2. Materials and Methods

IV.2.1. Study area

The study was conducted in the provinces of Sucumbíos and Orellana in the NEA (Figure 23A). It was restricted to lowland areas (retaining zones in ~144-900 m.a.s.l. in Amazon lowlands) that represent a surface area of 35,051 km^2 (Figure 23B). The area includes upstream oil and gas production activities potentially impacting forests, rivers and streams but also cities, villages and farming lands. This study excluded rivers with high flow rates (i.e., Napo, Tiputini, Coca, Payamino, Putumayo, Cuyabeno and Aguarico) where the surface waters and their interactions with groundwater are highly dynamic, therefore difficult to evaluate using scarce data. The study area is characterized by a super humid and warm climate and an average annual rainfall of 2900 mm (Institute of Meteorology and Hydrology-INHAMI). Hydrology regime is irregular with 1000 to 5000 m^3.s-1 daily discharges, and characterized by flash floods, due to extreme sensitivity to rain events (Laraque et al., 2007).

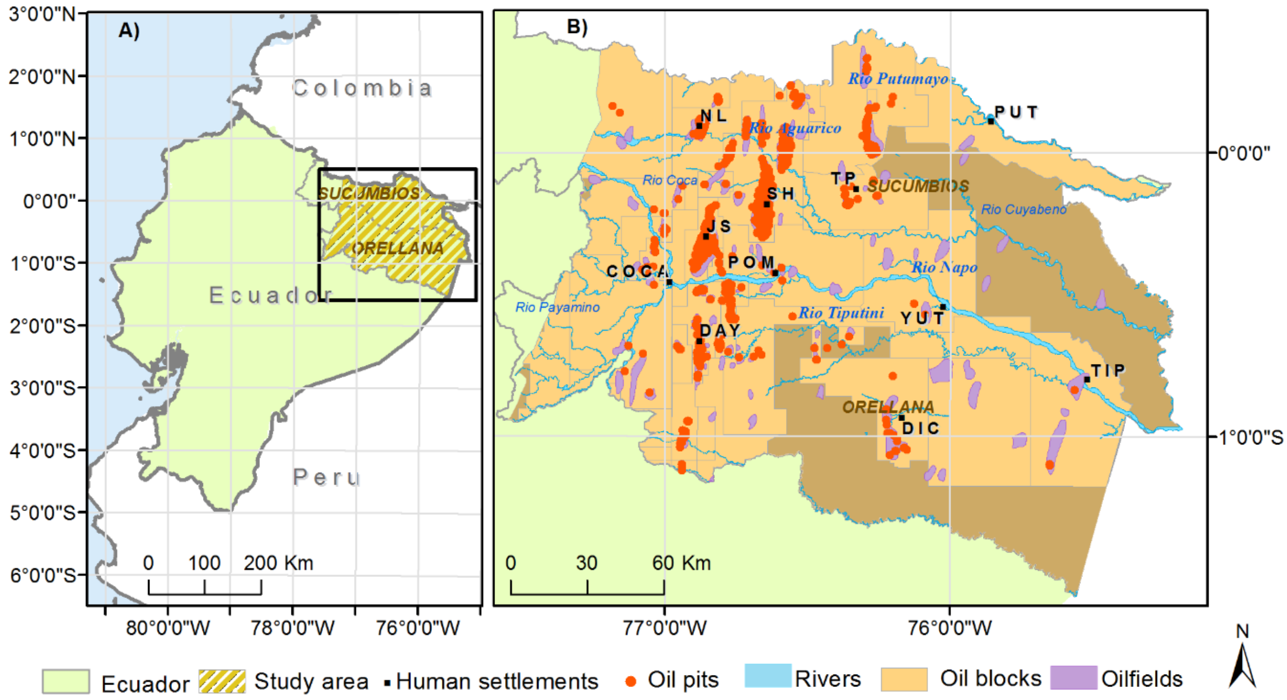


Figure 23. Location of unlined oil pits in the study area. A: NEA region; B: study area within the two NEA provinces of Sucumbíos and Orellana including oilfields and oil stations. Represented human settlements are abbreviated: DAY= Dayuma; NL = Nueva Loja (aka Lago Agrio); SH = Shushufindi; TP = Tarapoa; POM = Pompeya; PUT = Putumayo; COCA = Puerto Francisco de Orellana; JS = Joya de Los Sachas; YUT=Yuturi; DI=Dícaro; TIP=Tiputini.

IV.2.2. Compiling database for the modified DRASTIC index

Geographical data were collected from several sources over the study area, and have been classified into two categories: (1) intrinsic vulnerability, spatially evaluated by integrating available land use related spatial layers from local and national governments, which include: geopedology, hydrology, rainfall isohyets and a digital elevation model (DEM) publically available online at the National System of Information (http://sni.gob.ec/coberturas) and at the Geographical Military Institute' databases (http://www.geoportaligm.gob.ec); (2) Present-day hazard from pits, evaluated using available technical reports (Burbano et al., 2015; Cabrera, 2008; MAGAP-SIGTIERRAS and Tracasa-nipsa, 2015). TPH concentrations and pit volumes have been coupled with generic data. This data is summarized in Table 20. Spatial analysis were treated via ArcGIS® system.

Table 20. Summary of the data used for the different components of groundwater vulnerability and hazard.

Data used for Vulnerability	Component	Technical specifications	Sources and Date
Geopedology variables map	**D** depth to water **S** soil media **C** hydraulic conductivity - replaced by infiltration rate	1:25,000	Ministry of Agriculture, Livestock, Aquaculture and Fisheries (MAGAP) (1984) SIGTIERRAS* (2015)
Annual mean rainfall map	**R** net/final recharge	1:250,000	Ministry of Agriculture, Livestock, Aquaculture and Fisheries (MAGAP) (2002)
Hydrogeology map	**A** aquifer media, lithology and formations	1:250,000	Ministry of Agriculture, Livestock, Aquaculture and Fisheries (MAGAP) (2005)
Digital elevation model	**T** Topography, slope **R** hypsometry	1:50,000 90*x*90m resolution	Ecuadorian Military Geographic Institute (2010)
Soil types	**I** Impact of zone vadose	1:25,000	Ministry of Agriculture, Livestock, Aquaculture and Fisheries (MAGAP) (1984) SIGTIERRAS (2015)
Data used for Hazard			
Total petroleum hydrocarbons	Oil pits pollution content	EPA 1311 modified method	Amazon Defense Front (2008)
API density	TPH content estimation	Proof samples in several oil wells	National Board of Hydrocarbons (2014) Amazon Defense Front (2008)
Soil bulk density	TPH content estimation	Generic defaults	Environmental Protection Agency-OHIO (2005)
Soil texture	Soil bulk density	Derived from generic	Ministry of Agriculture, Livestock, Aquaculture and Fisheries (MAGAP) (1984) SIGTIERRAS (2015)
Oil pits	TPH content estimation and spatial location	Point shapefile Geometry	MAE-PRAS** (2014)
Digital elevation model	Cost distance	Raster layer	Ecuadorian Military Geographic Institute (2010)

* National System for Rural Land Information and Management and Technology Infrastructure
**Program of Social and Environmental Reparation by the Ecuadorian Ministry of Environment

IV.2.3. Intrinsic vulnerability: indexing and mapping

Intrinsic vulnerability was evaluated with spatial data layers collected by PRONAREG-ORSTOM (1984) and SIGTIERRAS-MAGAP (2015) technical field programs (Table 20) comprising physical-chemical information regarding soils in the NEA. Nonetheless, fieldwork areas from the second program excluded information in protected areas (Yasuní and Cuyabeno National parks). In these areas, data from 1984 were retained for parameters D, S, and I. Few "no-data" areas lacked values for soil characteristics in the data from the original table and were extrapolated from averaged values of same soil types from the 2015 dataset. In these areas, I parameter values were ascertained by the lithology from other soil types in the available dataset (Hamza et al., 2007). The parameters used are explained in sections IV.2.3.1, IV.2.3.2, and IV.2.3.3; and individual figures of complete rating, class, and weighting system are shown in Appendix B, section VIII. Subsequently, the two spatial datasets were merged into one coherent dataset.

Intrinsic vulnerability of groundwater was calculated via DRASTIC index modification (explained in detail in sections IV.2.3.1, IV. 2.3.2 and IV. 2.3.3), which is a standardized model built upon predefined weights, derived from expert knowledge and corresponding rating scores that may decrease or increase according to the contamination potential in a given area (Aller et al., 1987). Rating scores range between values of 1 for the less vulnerable, to 10 for the most vulnerable class. It is calculated using Equation 8:

$$DI = Dr\ x\ Dw + Rr\ x\ Rw + Ar\ x\ Aw + Sr\ x\ Sw + Tr\ x\ Tw + Ir\ x\ Iw + C\ x\ Cw \qquad \text{(Eq.8)}$$

Where,

D: depth to water table (*w* = 5),
R: net recharge (*w* = 4),
A: aquifer media (*w* = 3),
S: soil type (*w* = 2),
T: topography (*w* = 1),
I: impact to unsaturated zone (*w* = 3),
C: hydraulic conductivity (not available), use of infiltration rate (*w* = 5),
r: rating of area being evaluated
w: importance weight of the parameter

IV.2.3.1. Water-related parameters

Depth to water is a measure of depth from soil surface to the water table. A deeper water table level entails a lower probability for contamination potential: "Deeper the water table lesser the rating" (Aller et al., 1987). Some categories did not correspond to any parameter and were excluded from the analysis: sand banks and human settlements in spatial data for 1984. Spatial data in 2015 reported a water table depth item which was categorized as "not evident", it was assumed that potentially reaches tenths of meters depth or even more, that is deeper than the bottom surface of the pit, and this one was considered the less vulnerable class. The remaining categories varied from 0 to >1 m deep. Five classes were defined, and ratings scores from 1 -the less vulnerable- to 10 -the most vulnerable- were given (Fig. C1).

Final recharge represents the amount of water per unit area of land that can reach the water table. Recharge is mainly due to precipitation, which serves as pollutant transport. The greater the recharge the greater the contamination potential, and hence the rating value increases (Aller et al., 1987). The recharge was surrogated using a sub-model built on four weighted parameters, which integrate average annual rainfall, soil type, elevation and topography (Seabra et al., 2009). Elevation and slope data were extracted from the DEM produced by the Ecuadorian Geographical Military Institute (IGM) at a 1:50,000 scale. Although, the sub-model does not estimate a quantity for the recharge value, it allowed to index areas in terms of vulnerability categories (n=5) with score ratings from 1 to 9 (Fig C2.).

Topography was incorporated as water parameter because it plays a role in the recharge and water flow, even though it is also an independent parameter of the model. In this study, it is measured as slope percentage, using the DEM. The ratings are assigned following the premise that the less the percentage slope the higher the contamination potential due to remaining time of water and pollutant in a particular area (Seabra et al., 2009) (Fig C3).

Usually tables are used to show classes and ratings in this type of studies. In this study, long tables with categories or classes and ratings have been summarized with aid of maps instead.

IV.2.3.1.1. Hydrogeological settings and aquifer media

It refers to the saturated zone of soil, where the higher the permeability (larger grain size) and the lower the attenuation potential, the greater the contamination potential. The permeability variable is categorical. Six classes were identified overall, and subdivided into 11 categories among permeability and formation types, ranging from 1-10 in rating scores.

Alluvial deposits are given high ratings, located mostly in rivers banks and sediments. Medium scores of seven ratings are located in Arajuno, Chambira and Mena Fm, at western and center side of the NEA. Minimum ratings scores are given to Tena, Tiyuyacu and Chalcana FM

(Figure 24). An extended explanation of the hydrogeological formations can be found as supporting information (Fig C4).

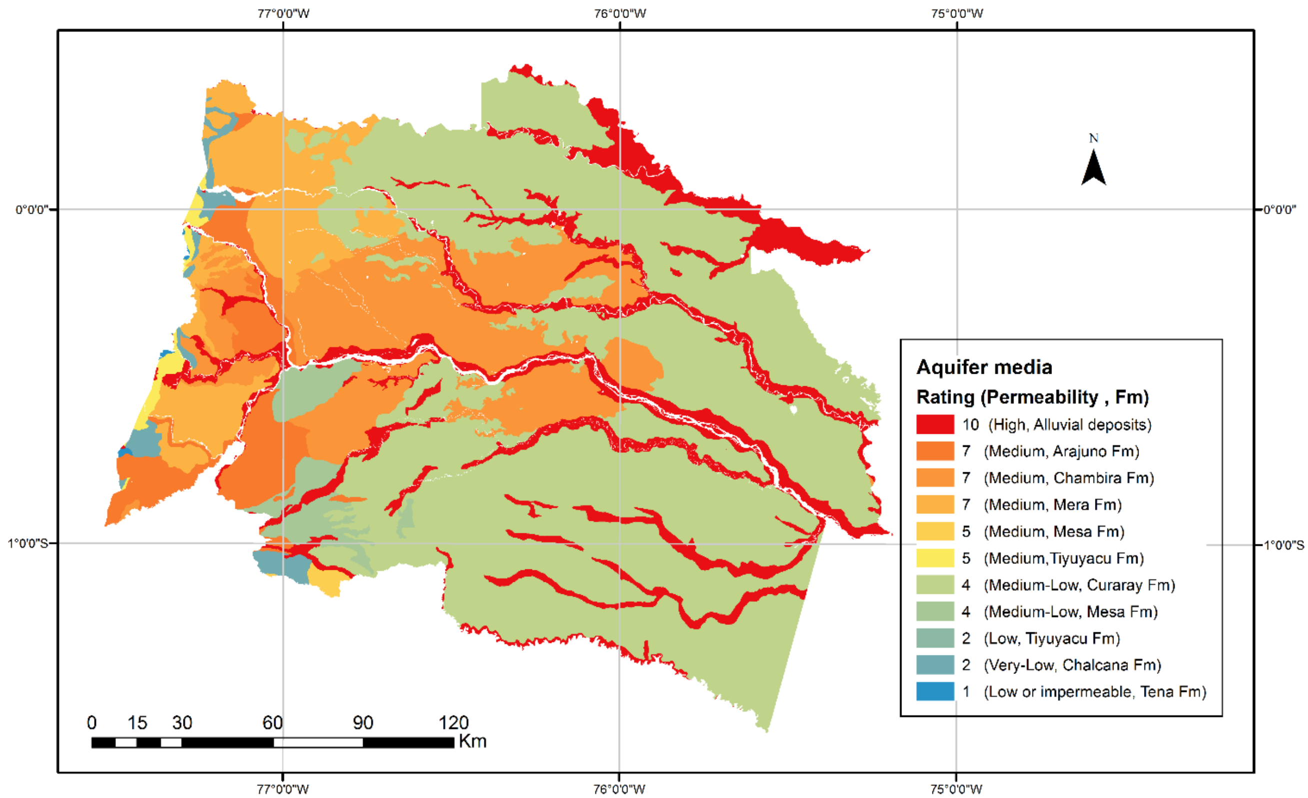


Figure 24. Hydrogeological settings and aquifer media depicting categories by permeability and corresponding given rating.

IV.2.3.2. Soil-related parameters

The soil parameters are summarized in: (1) soil texture, (2) impact of vadose zone, and (3) infiltration rate (which replaces the hydraulic conductivity) (Fig. C5). The three parameters are spatially corresponding one to another, in other words the resulting intermediate maps are very similar. Thus, in this study they are summarized into one map. However, it is worthwhile to define each one separately as follows:

(1) Indicates the uppermost portion of vadose zone, with significant biological activity. Also, the less the clay shrinks and swells, and smaller the grain, the less the contamination potential. The thicker is the grain, the higher is the permeability, and also the index rating.

(2) The impact of vadose zone is depicted by the unsaturated zone material. It controls and attenuates the passage of pollutants to the saturated zone. The thicker are the particles, the higher is the rating. Data from 1984 was used to define average values for null-data areas in 2015 GIS layers. The category soil texture at depth (measured within 50cm or more) is used to define the impact to the vadose zone, and superficial soil texture serves for rating in certain zones of protected area that lack information on this parameter (Hamza et al., 2007).

(3) The hydraulic conductivity, which is a lateral flow velocity of the water table, not available to the area, is replaced by the infiltration rate, which is defined by the vertical flow of rainfall through the soil surface column. This modification is plausible because depth to water is more than 10 meters in most areas of the NEA and maximum pit depth is 6 meters. One can consider that at this shallow depth, water flows in a vertical direction with discontinuities where there are subsurface and surface waters (Miguez-Macho and Fan, 2012).

Infiltration rates were of 5.1, 7.1 and 23.7 mm.hour^{-1} across the NEA region. These data aim to provide rating values to Soil texture and Impact to the vadose zone. They all were rated 3, 6 and 10 respectively.

IV.2.4. Sensitivity analysis

A sensitivity analysis was implemented to check the influence of parameter ratings and weights in the final index (Napolitano and Fabbri, 1996), to confirm whether or not a similar result output can be attained using fewer parameters, and to depict variations amongst them (Babiker et al., 2005; Kazakis and Voudouris, 2015). Two types of analysis were performed, the map removal analysis and the single parameter analysis (Napolitano and Fabbri, 1996; Rahman, 2008). The map removal analysis, which measures the sensitivity of the overall vulnerability index towards removing one layer, was computed using Equation 9:

$$S = \left(\left| \frac{V}{N} - \frac{V'}{n} \right| \div V \right) x\ 100 \qquad \text{(Eq.9)}$$

With: S the sensitivity value expressed in terms of percentage variation from the overall index, V and V' the originally computed and tested vulnerability indices respectively; and N and n the numbers of data layers used to compute V and V'. The originally computed vulnerability index represents the model using all parameters while the tested one, is the index resulting upon one parameter removal. Values closer to zero mean lower contribution to the overall vulnerability index; conversely, values greater than zero will indicate reduced contribution.

The single parameter analysis calculates the "real" weight of a parameter, expressed in percentage variation using only one parameter, incorporated in the calculation, and compares it with the "empirical" weight, given by the original DRASTIC model. The ratings of each parameter that could influence the model according to site-specific conditions is evaluated (Kazakis and Voudouris, 2015). It consists on selecting "unique condition subareas" (n = 300) of at least 10-pixels size and re-calculating DRASTIC index n-times upon variable rating scores, while the weights remain unchanged (Napolitano and Fabbri, 1996). The computations followed Equation 10:

$$W = ((\mathrm{P}r\ x\ \mathrm{P}w) \div V)\ x\ 100 \qquad \text{(Eq.10)}$$

Where: W the effective weight of each parameter; Pr = rating of the parameter and Pw = weight of the parameter, and V the vulnerability index as computed in expression Equation 8. For instance, the D parameter is multiplied by the corresponding weight = 5 and divided by the overall DRASTIC index.

IV.2.5. Hazard assessment: estimating volumetric properties of pits

Hazard can be defined as the probability of occurrence of an event that can harm a given environmental asset (Gleyze, 2002). In this case, the hazard potential is defined with petroleum related discharges, stored in the pits and eventually able to flow into groundwater. The materials found in these pits are numerous and their volumes are difficult to quantify/evaluate. Total petroleum hydrocarbons (TPH), usually required by legal international standards, were measured by the Amazon Defense Front (ADF) and were provided for hazard assessment in this study (Cabrera, 2008).

Oil pits have volumetric properties with technical model specifications well-established within the oil industry. Nonetheless, in the NEA, these pits have experienced several decades of physical alterations which have been registered by PRAS (2013). PRAS databases contain up-to-date volumetric dimensions, such as: volume (V), area (A), and depth (d) for 1,420 pits. Oil pits labeled as remediated/eliminated before 2001 were not considered. These dimensions were incomplete, so they were computed using a linear regression: *V=A x d (2,19m); lin.reg* (r^2 = *0.40)*. The median volume and area were extrapolated to pits with no registered information following the hypothesis that the documented pits are representative of all the pits.

At least, 87 oil pits were sampled (Table 21) for TPH content during a field work campaign (1995-1997) using the modified methods EPA 1311 and EPA 418.1 (Cabrera, 2008). The median TPH value was retained for extrapolation purposes. Summarized sampled data and volumetric dimensions collected in-situ are presented in Table 21.

Table 21. Descriptive statistics for pit volumetric dimensions and TPH contents. Statistic used for extrapolation to oil pits with missing data are in bold.

Statistic	**TPH ($mg.kg^{-1}$)**	**Area (m^2)**	**Volume (m^3)**	**Depth (m)**
Minimum	4	11.55	30	0.16
1st quartile	2,809	241.55	585.73	2
Median	**7,519**	500	**2,000**	2.19
3^{rd} quartile	21,749	977.92	2,000	3.3
Maximum	900,000	7114	18,432	10
Average	56,322	**835.7**	1,979.48	2.79
S.D.	170,472	914.48	2,163.1	2.66
Variation coefficient	3.03	1.09	1.09	0.95
Number of oil pits with missing data	1,333	144	1,200	1,298
Number of oil pits with data	87	1,276	220	122

IV.2.5.1. Estimation of TPH content in pits

In the NEA, several remediation processes have taken place historically. But some pits might have been insufficiently remediated by spreading soil without eliminating all residues, and others remain unlined (MAE-PRAS, 2016). Consequently, the NEA consists of pits with different substrates that represent a gradient from solid soil to oil-like media (Front of Amazon Defence, 2008), and need to be addressed first by calculating a TPH content in each pit derived from its volume. For doing so, a procedure should be implemented to incorporate the density of the substrate of the polluting source (Sarah et al., 2015). The TPH content in the oil pit was gradually fluctuating from oil to soil muddy mixture, difficult to measure for single oil pits, and useless at a regional scale. It was thus assumed a greater proportion (0.95) of each pit is represented by solid soil-like material and a minor proportion by liquid oil-like material (0.05). This assumption followed the ADF report, which assigned an attribute of "contained material within pits" to each sampled pit (Cabrera, 2008).

A density weighted Equation was implemented to estimate the TPH mass in each single pit:

$$TPH(kg) = \text{TPH } Q_2 \left[\frac{mg}{kg}\right] \times \left((\text{Sg}\rho \times 0.05 + P\text{b x } 0.95) \left[\frac{kg}{m^3}\right] \times V[m^3] \times 10^{-6} \left[\frac{kg}{mg}\right] \right) \quad (4)$$

Where,

TPH Q_2 = total petroleum hydrocarbons median value (7,519 mg.kg^{-1}) for extrapolating to pits and original values were used from the ADF sources; Pb = average soil bulk density values from literature (Saxton and Rawls, 2006) assigned to oil pits using the soil type layer that overlaps the pit; Sgρ = the specific density (Sg) calculated in each oil pit, using the expression Sg = 141.5. (API +131.5)$^{-1}$. Oil wells residues and hydrocarbons are discharged into pits. Pits are numbered according to belonging oil well, consequently the American Petroleum Index (API) density from the oil well was assigned to the expression (National Board of Hydrocarbons, 2016); V= oil pit volume (m^3) as calculated in section IV.2.7.

IV.2.5.2. Propagation of hydrocarbons at a regional scale

Pollutants transport depends on intrinsic parameters (DRASTIC parameters, groundwater velocity, transmissivity, initial water flow, water saturation, etc.) and factors related to hydrocarbon physicochemical characteristics and contamination sources (groundwater movement parameters like Darcy's and Bucket-Leverets 'equations, retardation coefficient, adsorption to soil, volatilisation fractions, oil-water relative permeability, cross sectional area, dynamic viscosity, etc.), usually implemented in a multiphase medium flow at site specific investigations (Fried et al., 1979; Kaasschieter, 1999; Ngo et al., 2014). In absence of detailed data for TPH behaviour and fate in the NEA, contaminating plume conception considers that pollutant flow rate is similar to that of water at a regional scale, therefore it is propagated from up-to-down slope flow direction.

Since the accurate fate of TPH transport into soils and waters at a regional scale is a difficult (if not impossible) task, the contamination plume derived from these aforementioned characteristics was mapped using different MDRs of maximum distance range potentially reached by water transported TPH (from 500 to 10,000 m) before pollutant is no mobile by water or ceases to cause harm to an asset is abbreviated by MDR.

IV.2.6. Contamination risk mapping: hazard overlaid with vulnerability

In this study, risk was calculated as a function of the probability of an acting hazardous source and the threat it represents (Gleyze, 2002; Metzger et al., 2008), as established in the Introduction of this thesis:

$$\text{Risk} = hazard \times vulnerability \qquad \text{(Eq.1)}$$

As prerequisite for applying Equation 1, hazard and vulnerability need to be spatially assessed first. The cost distance analysis was used to spatially evaluate the hazard. It calculates the least accumulative cost distance for each cell to the nearest source over a cost surface given a maximum distance (ESRI, 2015). The contamination from a pit depends on its TPH content and the downstream distance it can ultimately reach (input raster data). Indeed, the propagation from soil water leakage and runoff follows down the slopes (the cost surface) and its intensity decreases with the distance from the source. MDR were reclassified into classes from “low” to “high” from most distant to closest, considering that the hazard is greater with proximity to polluting source, because water leakage and runoff are the assumed major dispersion processes. The risk for groundwater was calculated multiplying the hazard propagation for each distance threshold with the vulnerability map, also reclassified from “low to “high” in a pixel-wise approach. The risk map obtained had “low” to “high” classes, as well. Four MDRs with contrasted risk spatial patterns were selected to feed the discussion after several MDR were explored.

IV.3. Results

IV.3.1. Influence of single groundwater parameters to overall vulnerability

The rating scores or classes, represent the influence or contribution, of each of the seven parameters to the whole DRASTIC index to define the vulnerability across the NEA, and are summarized in Figure 25. Rating scores are one factor determining vulnerability that considers the NEA specific conditions, the other factor is composed by weights from the empirical/adapted model. The parameters contributing to vulnerability are explained from highest to lowest weight:

Depth to water (W=5) indicates that >10m depth was less rated class, corresponding to 36% of the NEA region, followed by medium vulnerable classes 0.5-1m class with 15% and highly rated shallow waters 0.0-0.5m accounting for 12% along Cuyabeno protected area, the Napo and Putumayo River in the most eastern side of the NEA (Figure 25).

The influence of final recharge (W=4), greater in the Andean foothills, yet accounts for only 5.5% of total area; the southern part of the Napo River, north side from Dícaro and majority of Orellana province had lower ratings accounting for 21%, and medium ratings for about 27% of area in north from Napo River along the Tiputini River (Figure 25).

The surface of the soil media group (each parameters empirical weight correspond to soil type W=2, vadose zone W=3, infiltration rate W=5) contributed to 60% of low, 27% medium and 9% of high ratings. Soil media spatially overlays with infiltration rate and impact to the vadose zone, having high rating scores from northwest (NW) towards southeast (SE), notably along Nueva Loja, Shushufindi and some parts on NW and SW towards the Andean foothills (Figure 25).

The soil media perimeters show a higher rating scores of 10 attributed in Nueva Loja, Shushufindi, and Coca cities in the NW side of the NEA occupying an important surface of the total study area (Figure 25).

The aquifer media (W=3) shows higher rating levels (corresponding to high permeability) on the riverbanks and alluvial deposits with 12.7% of surface, medium permeability zones are found in 54.7%, and low permeability in 29.2% of the entire NEA region (Figure 25).

Topography (W=1) shows highest ratings in the Cuyabeno reserve as well as in Joya de Los Sachas parish. Highest ratings corresponded to a 55.6% of total area (0-12% slope), and medium rating scores in the rest of the area (12-25% slope, for 38.2% of surface) (Figure 25).

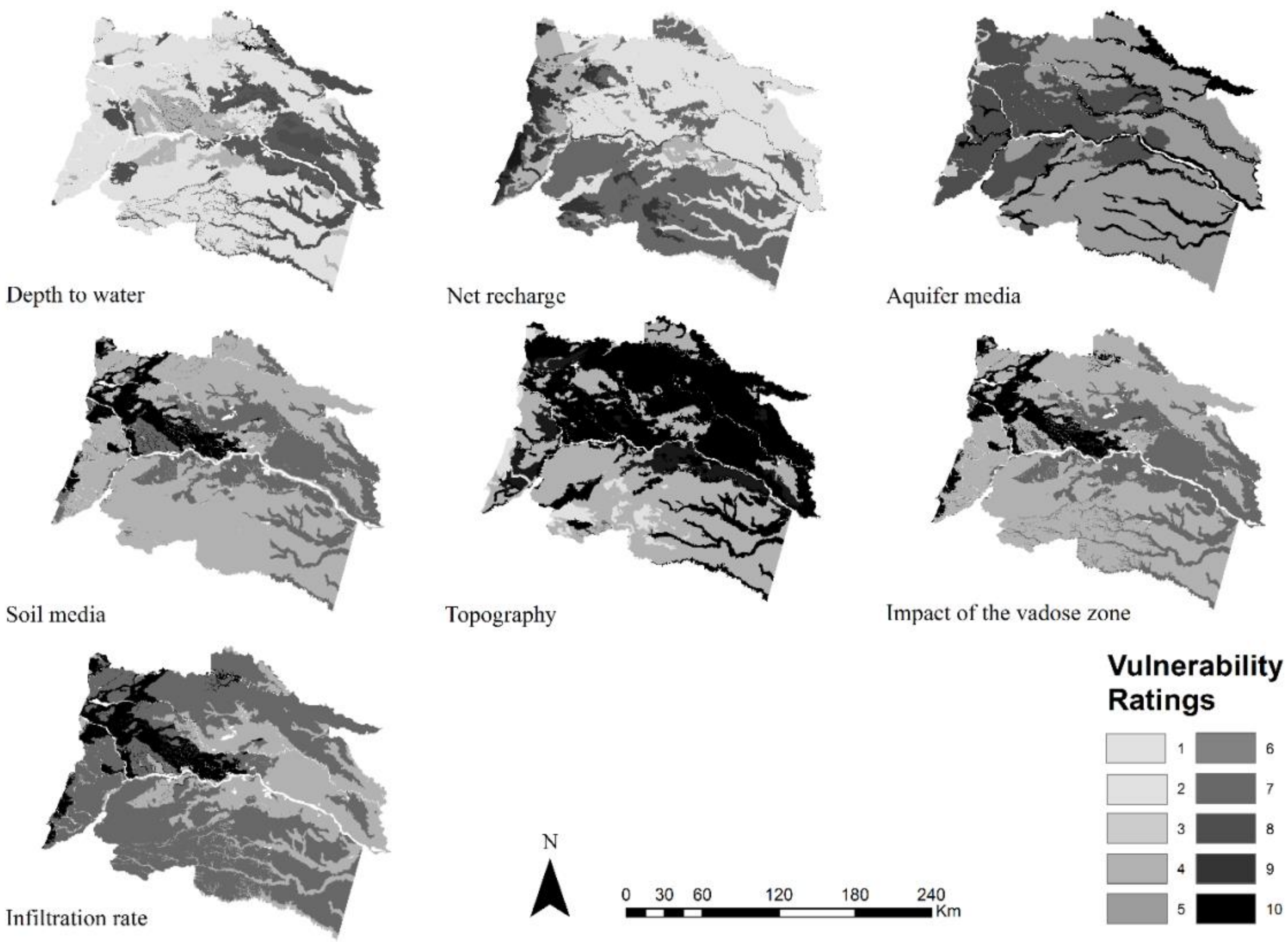


Figure 25. Vulnerability ratings scores obtained with the seven parameters for the modified DRASTIC model.

The overall vulnerability index depends on the rating scores of each parameter multiplied by the weights defined by the model. Thus, the result of rating scores on the vulnerability index differs accordingly. The resulted vulnerability index indicates that the three medium vulnerability classes are of similar surface area while low and high vulnerability classes correspond to equal smaller surfaces (Figure 26.).

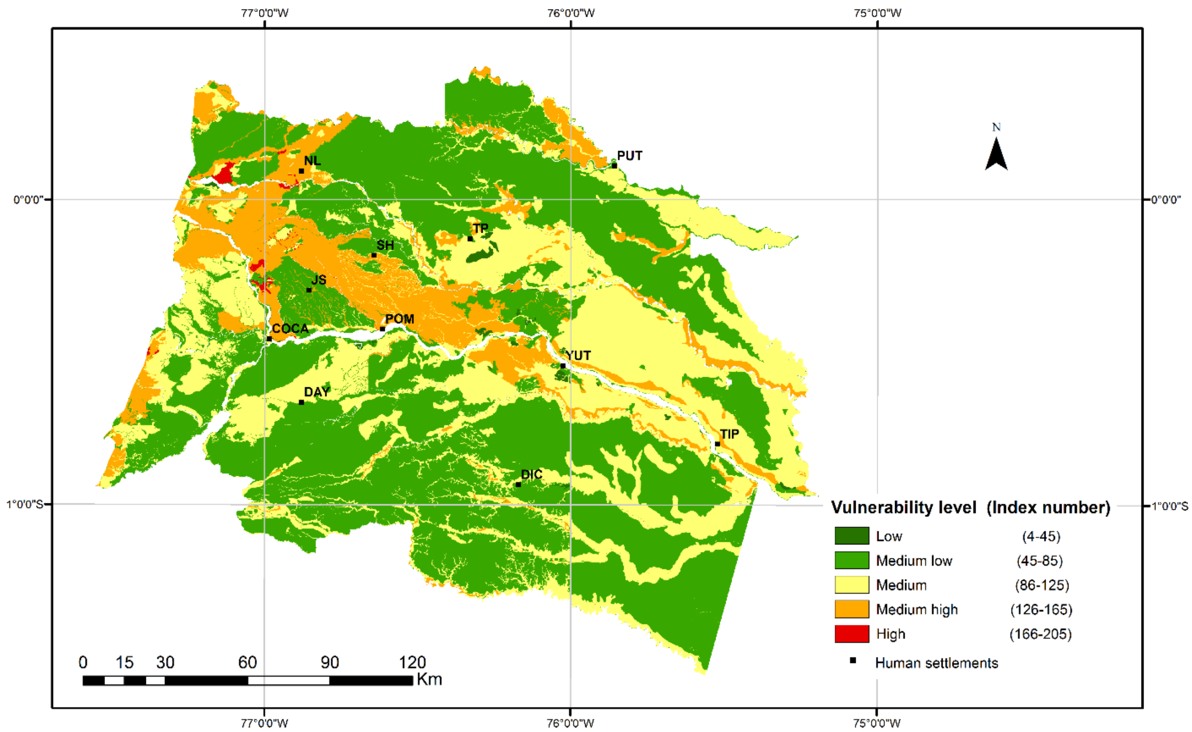


Figure 26. Intrinsic vulnerability index values and corresponding categories for the entire study area. Surface areas corresponding to high, medium high, medium, medium low and low vulnerabilities represent respectively 0.25% (90 km^2), 15.3% (4,650.4 km^2), 31% (11,110 km^2), 54% (19,048 km^2) and 0.4% (141.4 km^2). Nueva Loja, Shushufindi, Coca, and Pompeya are in the highest vulnerable zones, followed by Tarapoa and Tiputini. Levels are classified using equal intervals in vulnerability indices.

IV.3.2. Impact of parameter removal and assigned weights on vulnerability index variations

Table 22 shows variations in the vulnerability index using the *single map removal* approach. The variation is the highest upon removal of final recharge parameter (mean variation in vulnerability index = 2.14%) and less sensitive to depth to water, yet important and also high in its weight. The impact of vadose zone and infiltration rate seem to account less to overall vulnerability index. The least average variation is found when removing soil media and topography.

Table 22. Single map removal sensitivity analysis of the DRASTIC index. Influence of the various parameters on DRASTIC index variations according to the single map removal sensitivity analysis (SD: standard deviation).

Parameter removed	Variation in vulnerability index (%)			
	Mean	Minimum	Maximum	SD
Depth to water (D)	1.9	0	5.6	1.6
Net Recharge (R)	2.1	0	13.9	1.1
Aquifer media (A)	0.8	0	14.3	1.2
Soil media (S)	0.004	0	2.4	0.1
Topography (T)	0.1	0	5	0.1
Impact on saturated zone (I)	1.2	0	7	1.2
Infiltration rate (C)	1.3	0	7.1	0.8

The effective weight mean indicates the departure extent from the weights given by the empirical DRASTIC model according to the subareas specific vulnerability conditions (n for V' = 300). Accordingly, the aquifer media, infiltration rate and topography have higher effective weight means, showing the importance of obtaining more detailed data on these parameters. Typical higher weighted parameters such as Depth to water and final Recharge were assigned less effective weights (Table 23). Topography is given a low empirical weight of 1, which demonstrates its low contribution to the DRASTIC index, yet it is also used to calculate the final recharge. This suggest its effective weight on the overall index could be higher than expected.

Table 23. Statistics of the single parameter sensitivity analysis.

Parameter	Empirical weight	Empirical Weight (%)	Effective weight (%)				
			Mean	Minimum	Maximum	SD	Redefined weights
D	5	21.7	13.5	2.8	48.1	11.6	3.1
R	4	17.4	15.2	2.2	90	11.8	3.5
A	3	13	17.6	2.1	100	6.7	4
S	2	8.7	8.5	3.2	22.2	2.6	2
T	1	4.4	7	0.6	62.5	4.3	1.6
I	5	21.7	21.5	9.1	52.6	6.1	4.9
C	3	13	17.1	5.2	37.5	6.8	3.9

IV.3.3. Risk mapping: spreading contamination in groundwater vulnerability zones

The cost distance analysis provides a maximum distance potentially covered by the contamination plume which describes the pollutant transport from the source. This procedure results in as many maps as possible. In this study, four risk maps of potential MDR based on input TPH content that simulated the risk plume were retained (Figures 27A to 27D). Figure 27 shows that the higher the maximum distance, the higher the surface potentially impacted by the contamination. Moreover, Table 24 shows that the shorter the maximum distance, the larger the area concerned by highest risks.

The surface extent change needs to be considered according to the different levels of proposed risks in function of the downslope and flow direction of the local territory. For instance, the highest risk (red coloured on maps) decreased, its area from 19.19% to 2.22% of the total surface at risk, but low to moderate MDR indicate increasing from 14.3 and 59.4 to 42.3 and 30.7 surface of the total risk zone, respectively. The "medium distances" of 2,000 and 5,000 meters

(Figure 27B and 27C) have greater surface at moderate and low moderate risk area surface in relation to total risk area than other categories of risk.

The “closest distance” of 500 meters (Figure 27A) indicates highest risk contamination hotspots are found in Nueva Loja and Tarapoa, medium to high risk zones dispersed nearby Shushufindi and of Joya de los Sachas. However in the intermediate MDR the hotspot are remarked in south Shushufindi Tarapoa, north Joya de los Sachas and east of El Coca, as well.

According to the “longest flow distance” of 10,000 meters TPH may spread over 28% of the NEA surface. Low to moderate risk of contamination is connecting the hotspots of risk previously defined in the 2,000 and 5,000 meters, except for the risk zones at Dícaro and east form Yuturi, which are oil blocks with road access restrictions. The surface extent of the risk zones for the four maps are depicted in Table 24.

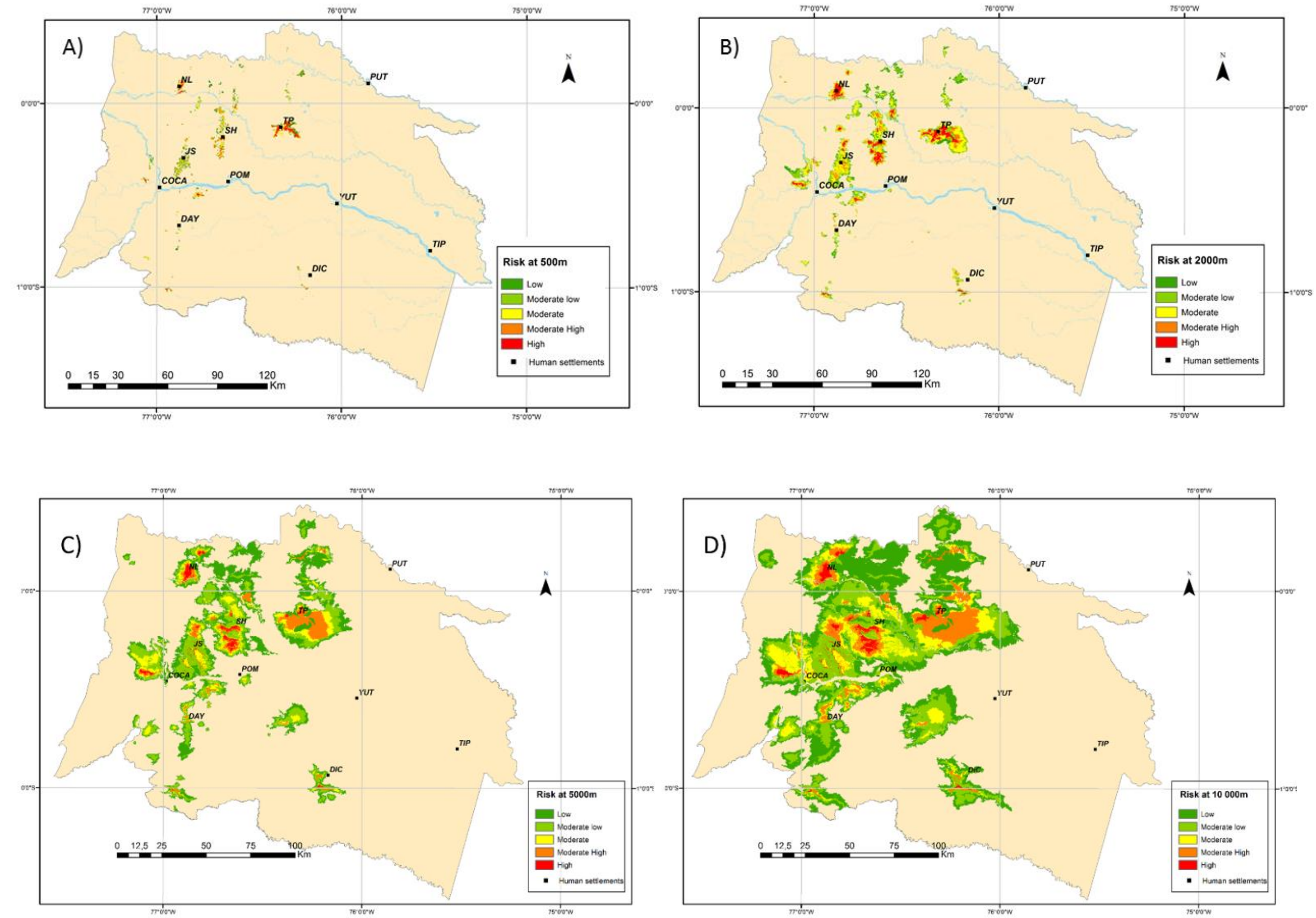


Figure 27. Risk map of TPH discharge from pits into groundwater as a function of maximum distance range of flow proposed: A) closest distance to source; B) and C) intermediate distance flow; and D) longest path flow of contamination.

Table 24. Surface areas (in %) corresponding to each risk level, depending on the maximum distance of plume. The figures illustrating each risk MDR are indicated. Δ = change of the surface at risk based on potential MDRs.

	Distance flow (m)	**500**	**2,000 (Δ)**	**5,000 (Δ)**	**10,000 (Δ)**
Risk level (km^2)	Low	48.6	253.2 (204.6)	1,992.9 (1,739.7)	4,229.7 (2,236.8)
	Moderate low	100.0	393.3 (293.3)	1,418.4 (1,025.1)	3,070.9 (1,652.5)
	Moderate	92.6	446.8(112.6)	559.4 (112.6)	1,423,9 (864.5)
	Moderate high	34.3	165.1 (130.8)	461,8 (296.7)	1,063.9 (602.1)
	High	65.5	174.7 (109.2)	110,6 (-64.1)	222,3 (117.7)
Total area at risk (km^2)		341	1,433.1	4,543.1	10,010.7
Figure		25A	25B	25C	25D

IV.4. Discussion

IV.4.1. Components of risk: vulnerability and hazard

IV.4.1.1. Groundwater vulnerability

Overall vulnerability was medium low which correspond well to the general argillaceous soil medium conditions of the Amazonian soils. The results indicate that the groundwater vulnerability assessment is robust. The *map removal* sensitivity analysis indicates a relatively low maximum mean variation of 2.1% for a given parameter, while other studies showed significantly higher values of 15.1% (Babiker et al., 2005), 13.7% (Rahman, 2008), or 7.7 % (Zghibi et al., 2016). In all cases the net recharge had the highest mean. The *single–parameter* sensitivity analysis resulted also in lower effective weights with higher variations, perhaps because of the aquifer formation differences between sandstone (Curaray) and quartz pebble bearing (Chambira) matrices. These differences reveal the adaptation capabilities of the DRASTIC model depending on the aquifer matrix (Kazakis and Voudouris, 2015), covering the majority of the argillaceous NEA surface. Likewise, this argillaceous matrix explains the extensive medium-low vulnerability class due to reduced permeability. The most representative parameters impacting vulnerability are aquifer media, topography and infiltration rate which is in agreement with the sensitivity analysis performed in a previous study (Babiker et al., 2005). The aquifer media inflicts the largest impact on groundwater vulnerability according to the map removal sensitivity analysis, while the single parameter removal indicates that infiltration rate explains 76% of the potential information loss, probably because of not integrating hydraulic conductivity (effective weight *vs.* empirical weight ratio of 0.76). These aforementioned explanations of the model adaptations suggest that integrating up to present available quantitative data is constructive. Vulnerability assessments using the DRASTIC model have already been applied without integrating all parameters, notably hydraulic conductivity, with satisfactory results (Al-Adamat et al., 2003).

IV.4.1.2. Impact of hazard assessment in overall risk

The total TPH content in oil pits within the NEA resulted in 49,436.4 t adding the original (ADF data) and the calculated amounts (extrapolated data). These amounts resulted in 10.2 less than crude oils discharges, according to latest official data of 505,600 t (MAE-PRAS, 2016). The difference in part may be explained because official data amounts are crude oils while this study considers only hydrocarbons but the difference is significant. Then, TPH concentrations were implemented to assess hazard, while other studies restrain only to model hazards from land use (Kazakis and Voudouris, 2015; Mimi and Assi, 2009). The portion of pits having TPH contents (6%) and volume characteristics, i.e., surface (92%), volume (15%), depth (8%) suggests that, at regional scale, hazard assessment accuracy might benefit from continued field data collection (MAE-PRAS, 2016). The cost distance analysis rendered higher spatial resolution maps than previous studies that were performed only at the parish level, i.e., epidemiological studies (Hurtig and San Sebastián, 2002). Subsequently, the combination of hazard and vulnerability maps enabled mapping MDR for regional land use planning. Worth mentioning is that OP also discharge produced water (3:1 water to oil ratio globally estimated) containing mobile radioactive metals (Jerez Vegueria et al., 2002). Although, these data were not available in the NEA for suitable estimation, the MDR plume would potentially represents produced water hazard as well.

IV.4.2. Land use planning implications

MDR mapping are useful for decision-makers to anticipate for potential environmental impacts (Lahr and Kooistra, 2010). The analysis on cost distance reveals that while the area at risk increases, the proportion of higher risk area decreases, which means that if the contaminant is not labile, it will affect a smaller area but with higher TPH concentrations. The delta obtained from the different MDR indicates that the proportion of low to moderate-low levels of risk increases and that the proportion of moderate-high levels remains similar across all MDR (Table 24). This might be occurring because the different micro slopes throughout the Amazonian basin act as barriers but also due to intrinsic properties of the algorithm used. In addition, the surface area at risk depends highly on the vulnerability map and less on the number of pits. It seems that non-significant difference is obtained using coupled original + extrapolated TPH content from using only volume of pits without pollutant content (Fig C7) when comparing the respective figures (Figure 27). According to this, then the TPH sampling errors coming from the original dataset could be considered of reduced importance and more relevance should be considered upon volumes of the source.

Figure 25A might display a current state of the risk of contamination from older pits, where only the heaviest fractions remained, low to medium molecular weight hydrocarbon fractions might have potentially reached MDR and discharges have ceased owing management improvements (pits have been lined up, no new pit has been built) (Fig C6 and C7).

Although TPH samples were evaluated under EPA 1311 and EPA 418.1 modified methods (Cabrera, 2008), they have been internationally criticized because of lack of scientific rigor (Buccina et al., 2013) . Spatial planners and local decision-makers can use locally collected data for hazard assessment when certain criteria is met, i.e., social trust on local regulatory institutions, recognition of local communities, post-hoc attempts for global recognition of environmental impacts during Texaco operations (Bronfman et al., 2009; Gómez, 2013; Otieno-odawa and Kaseje, 2014; Schweitzer, 2008). The 500m to 2,000m MDR better provide insights for spatial planning of human activities, such as agriculture or drinking water, that could impact local public health if improperly planned (Lahr and Kooistra, 2010). This is because most likely on the other MDRs the biological and microbial degradation (Maddela et al., 2016; Singh and Chandra, 2014), organic matter content (MAGAP-SIGTIERRAS and Tracasa-nipsa, 2015) and hydrocarbons adsorption to soil influence the MDR capacity in Amazonian environmental conditions. In spite of inconclusive results about disease incidence correlated to oil pollution, i.e. cancer incidence, controversies remain in official health data regarding spatial exposure to oil activities (Hurtig and San Sebastián, 2002; Moolgavkar et al., 2014). However, it is advisable to communicate about the potential of pollution in drinking water from wells in highest and medium high risk areas, i.e., zones overlapping indigenous communities because cancer incidence rate has been 63% higher than for immigrants (Kuang-Yao Pan et al., 2010). In contrast, in most probable MDRs (Figure 27A and Figure 27B) the populations that inhabit these regions are mostly immigrants or colonists, except for Tarapoa and Dícaro. Highest risk zones, e.g., in 500 m MDR should be considered with special attention for agriculture, specifically cacao plantations whose grain quality might be affected by cadmium intake (Barraza et al., 2017)

IV.4.3. Limitations and further adaptations

Sensitivity analysis indicates the importance of each parameter on the final results, by removing them or adjusting weights to the study site specificities (Kazakis and Voudouris, 2015; Napolitano and Fabbri, 1996; Rahman, 2008). The significant geographical overlapping of S, I and C (the soil-related group) indicates single parameters' weight is low (Fig. C4). Nonetheless, results suggest that total vulnerability is affected by the combination of these parameters (added up to the highest cumulative mean variation with a mean index of 2.5 %). These are similar sub-surface texture zones. These parameters are spatially co-dependent on soil types, thus more detailed data on these parameters is required.

The single map removal analysis suggests that weighting factors are more relevant to define vulnerability than the rating scores. For instance, the variation of vulnerability index is highest by removing final recharge (2.1%), and might be attributed to higher empirical weights (R=4) and not to the high rating score, contrary to other studies (Babiker et al., 2005). This effect is even clearer for the D parameter, where the different rating scores could be simplified to one or two i.e. less than 1 meter and deeper. Current data used to rank this parameter lack detailed explanations in some zones that indicated values of "considerable" depth to water, probably not detected due to the use of shallow monitoring systems (MAGAP-SIGTIERRAS and Tracasa-nipsa, 2015).

The accuracy of several vulnerability parameters such as infiltration rate and aquifer media need to be improved. The effective weight variation analysis shows that these parameters are weighted higher than the empirical 4 and 3.9, respectively. Maybe the replacement of hydraulic conductivity by infiltration rate is responsible for this change and reduced accuracy. One of its components, which is soil thickness (Fig C4) is already available; the other, transmissivity, is not (Hamza et al., 2007; Kazakis and Voudouris, 2015).

Certainly, many variables and parameters need better accuracy, completeness in Amazonian regions, and across Ecuadorian landscapes (Ribeiro et al., 2017), that may be key to improve knowledge of groundwater behavior. Only four aquifers have rigorously been identified in the NEA, which represent less than 2% of the total study area. Aquifers in the Napo basin showed variable average flow rates from 0.89 to 13 $ml.s^{-1}$. This knowledge is expected to improve in the near future as more research is currently being done (SENAGUA-ESPOL, 2017).

In addition, topography is a dependent variable, used for different computations, such as final recharge and hazard propagation, and its importance might be overestimated. More precise data on these parameters may reduce redundancy on parameterization, even if its redundancy is inherent to the design of the DRASTIC model (Gogu and Dassargues, 2000; Merchant, 1994; Secunda et al., 1998). Nonetheless, this study endeavored to use a modified DRASTIC rather than other methods that use less parameters ,*i.e.*, susceptibility index (SI) (Ribeiro et al., 2017), GOD or EPIK methods, because it is more informative (Merchant, 1994; Secunda et al., 1998; Shrestha et al., 2017).

More precise data for understanding the specific molecules represented within TPH’s and their fate in tropical soils and their toxicity is needed in the NEA. Other objectives such as selecting sites for detailed further research (finer spatial scales) can be derived from selecting pits at different risk zones. As data become available for assessing intrinsic vulnerability factors (*i.e.* porosity, hydraulic conductivity and gradient, groundwater velocity (Kazakis and Voudouris, 2015)) and factors for physico-chemical properties of contaminants (*i.e.* cross-sectional area, partition fractions, biodegradation, sedimentation or chemical complexion) a multi-phase flow approach using in situ measurements would benefit from the use of specialized software (*e.g.* COMSOL, Modflow, Hydrus-1D, FEFLOW, etc.) (Ngo et al., 2014) integrating stochastic techniques, i.e., Markov chain model (Weigand et al., 2001). It would be informative to apply such models on OPs in directly downstream the monitored area; for instance, to evaluate probabilities of TPH leaching from OP.

IV.5. Conclusion

This study is the first spatial assessment of oil pits-associated risks to groundwater proposed for integration in local water and land use planning and as a decision making tool for local and national regulation and for installed oil and gas operators. This spatial risk assessment provides insight by proposing MDRs with variable maximum distances for hydrocarbon propagation. Several potential sampling sites can be delineated for further research on hydrocarbons behavior based on a combination of pit locations within different risk level zones (Ribeiro et al., 2017). Risk should also be linked to other human activities and social variables, i.e., overlaying with land use and occupational maps (Al-Adamat et al., 2003; Rahman, 2008; Secunda et al., 1998) This study maps the potential contamination of groundwater from pits, having not being remediated for most of them, or at least largely insufficiently. It is built on large spatial datasets, physicochemical properties of soil and hydrocarbon characteristics, to improve the contamination risk assessment to aquifers. Future research should incorporate an improved approach, via the use of refined weights, and the gather of data to improve groundwater parameters understanding. Local decision-makers and spatial planners may avoid potential adverse impacts within high risk zones at 500 or 2000m MDRs by encouraging further investigations in these areas, including validation of contaminant plumes. Land use plans should revise for potential relocation of activities in high risk areas designated for agriculture, increase potable water service and intensify risk communication, *e.g.* improve social perception of industrial activity in the NEA by integrating disclosures on practices and quantifiable risks to colonists and indigenous groups for improving public health.

Chapter V - General discussion

V.1. The significance of the risk assessment approach

This thesis presents a spatial risk assessment encompassing the evaluation of its two components, i.e., hazard and vulnerability. Spatial risk assessment is significant as a method for minimizing gaps in knowledge through the integration of a large number of heterogeneous data that can be indexed. Its significance lies also in that it can integrate multiple different datasets from many institutions. This is much so because at a regional spatial scale it helps find areas where more "in situ" information will be needed. A pre-requisite is that it needs to be able to represent or aim to advance in a manner that it allows the independent assessment of hazard and vulnerability.

In this sense, the first two chapters advance towards hazards assessments which, while not explicitly considering exposure effects, indicate the valuable potential that long-term emissions represent for hazard assessment. Chapter III and IV deal with vulnerability assessment and even risk assessment. Specifically, the conceptual framework for this approach was presented to help assess spatially the risk of contamination from oil activities in the North-eastern Ecuadorian Amazon (NEA), at a regional scale. It was framed within the research program named MONOIL which was granted by the "Agence Nationale de la Recherche" (ANR) with organizational and logistic support of the "Institute National de la Recherche pour le Developement" (IRD). The NEA is one of the impoverished areas of Ecuador, encompassing great biological and cultural diversity, where habitants allege not to significantly benefit from crude oil production and its revenues.

V.2. Spatial emission inventories

First, in order to assess hazard, the discharges from oil activities at the pollution sources were estimated; which is represented in six maps that indicate spatial locations of these sources and long-term emission spanning 11 years. Spatial inventories at infrastructures were mapped and could be classified into three categories:

<u>Oil spills:</u>

- Infrastructure relevant to oil spill occurrence within well- and poorly-documented oil blocks.
- Assemblage of oil spills reported originally by Ecuadorian institutions.

- Homogenization of oil spills by combining the assemblage of oil spills reported and the probable oil spills discharged in the poorly-reported oil blocks across the NEA, using the spill rate factor determined on well-reported blocks.

<u>Gas flaring and venting:</u>

- Infrastructure potentially releasing gas due to flaring process, and oil fields with APG and without data.
- Black carbon emission estimates.

<u>Unlined oil pits:</u>

- Total hydrocarbon contents
- Age of pits as documented by PRAS (Figure C6)

Several negative and positive interpretations could be drawn from these maps:

First, the grid cell size might be of a lesser size. Studies have revealed ecotoxicological impacts at less than 2 km of pollutant sources (Dung et al., 2008; Soltanieh et al., 2016; Waldner, 2008; Wernersson, 2004). A positive aspect, as stated in Chapter II, is that mapping hazard at 25 km^2 represents and advancement relative to regional or global models that are currently available (Freitas et al., 2009). This aspect, added to the fact that wind direction is equally changeable, might suppose that in general the cell can represent an averaged hazard plume. Besides, at this grid-cell size, errors about geolocation of emissions are of reduced importance. These spatial inventories have the potential to be input into transport modelling software in future studies, to map contamination plumes more precisely, which could be coupled with specific exposure data to assess further important societal assets, such as human health. Some authors argue that mapping for long-term exposure to pollution could also represent a surrogate for hazard (Barraza et al., 2017; De Miranda et al., 2012).

In addition, studies have assessed oil-related hazard implementing the age of infrastructure (Fig C6), because it is often considered that the older the infrastructure, the higher the probability of failure due to maintenance issue (corrosion, restraint budget to repair and supply newer technologies). Contrary to these studies, in this thesis the age of infrastructure did not provide insights that could help integrating into future models.

This thesis presents maps that attempt to provide regional representations of hazardous emissions from oil activities throughout the study area, organized in two groups of elements: (i) the infrastructure and its spatial distribution over the territory; (ii) the spatial emission inventories. In the case of oil spills, well- and badly-reported blocks are mapped under two different hypotheses. There is a first scenario, where the original data is considered well reported by public institutions. It is retained as a *ceteris paribus* situation, where the institutions are reliably reporting and disclosing spill data; in this scenario, the significant differences regarding spill occurrence across the blocks (at a given infrastructure density) would reflect different management procedures or technologies implemented. The second scenario considers that there is no reason why the occurrence of an oil spill should vary significantly across the blocks, at a given infrastructure density, which allows to identify some blocks as presumably poorly reported. In this case, spill occurrence is not in agreement with a spill probability that would rely only on the currently present infrastructure. The situation of disclosure is considered as not equally reliable according to the blocks in this scenario. As the two scenarios are not mutually exclusive, oil spills that are actually occurring might range between the two maps. Homogeneity of spills serve to provide an alternative approach for hazard mapping because it relies more on actual spill data than on infrastructure density. It comes interesting to compare these maps with the map of infrastructure density. Evidently, the densities by watershed can provide another picture of real situations and it is easier to implement (Fig D1), but is not necessarily meaningful for flare stacks that release airborne emissions. However, the case of airborne emissions is less complex to deal with, since the data can be compared to other datasets, including satellite products, deemed as more reliable, because less prone to external manipulation and disclosure hindrances. The quantification of the emissions as inventory can improve safety and security planning, and enhance public institution accountability.

V.3. From the general concept of vulnerability to its ecological specificities

Vulnerability is a term that has been increasingly used in scientific research in the latest years. In the introduction, its relation to climate change and social context for development were explained, as well as its roots in the field of ecotoxicology. The intricate concepts and their overlapping sometimes show the difficulties to establish a systematic process for its evaluation, even within the environmental side of vulnerability. This issue is revealed by the lack of consensus over its definitions, as shown in the literature, which resulted in interchangeable terms with similar operational use: e.g. environmental, ecosystem and ecological vulnerability. Environmental vulnerability is considered as the susceptibility of the natural environment to negative impacts of events or processes (Beroya-Eitner, 2016; Flax et al., 2002). The environment is defined by physical, chemical, biological, geological and geographical conditions, including the atmosphere, land, water, plants and animals, humans and their interactions. Ecological vulnerability is defined as the potential of an ecosystem to regulate its response to a stress, which occurs at various time and space scales and includes also the common components of susceptibility, exposure, and resilience (De Lange et al., 2009; Turner et al., 2003). This potential is determined by the characteristics of an ecosystem that contains different hierarchical levels, i.e., organisms, populations, communities, habitats and ecosystems (De Lange et al., 2010). Hence the consideration of the Potential Ecological and Biodiversity Vulnerability Index (PEB), referring to the ecological integrity and biodiversity at a given moment, agrees with the latter concept, where the human is not considered as part of the system but as a potential hazard. The use of dispersal models of species, or mapping habitats at higher spatial resolutions, could provide more parameters to develop more complete indicators for ecological vulnerability. In addition, the same type of evaluation could be integrated at different time scales, using historical land use changes, coupled with the several changes of the protected area boundaries. An example is portrayed in (Fig) that indicates the changes over time in the boundaries of Yasuní national park.

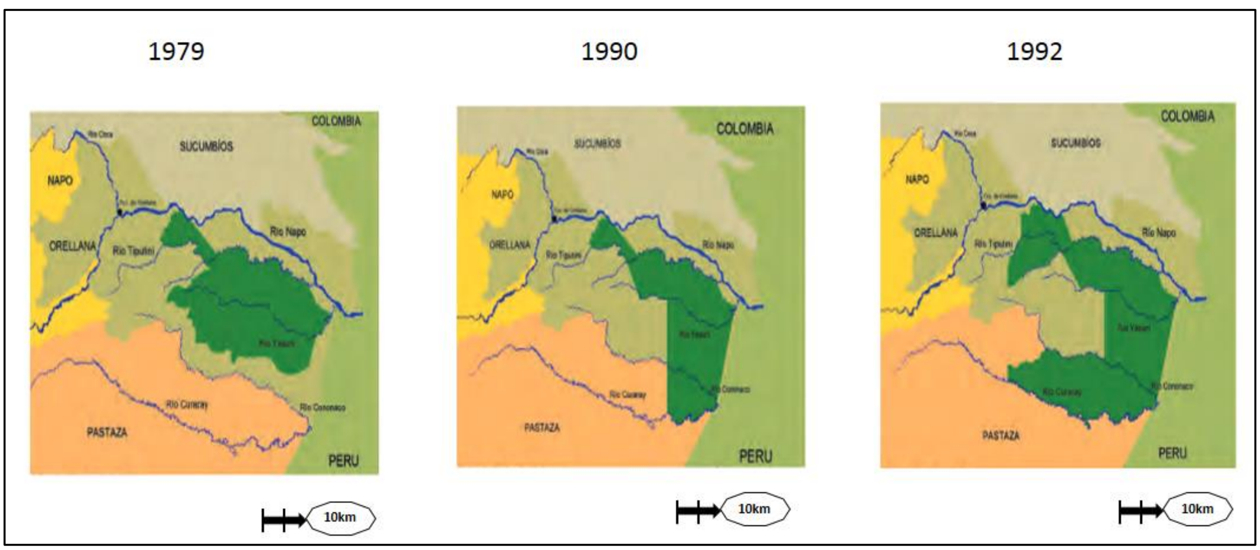


Figure 28. Historical shifts of legal boundaries of the Yasuní National Park (Fontaine and Narvaez, 2007).

The assessment of ecological vulnerability requires consideration for time and space scales, and their hierarchisation. It includes both the evaluation of ecosystem functioning and the relationships between abiotic environment and living organisms (Beroya-Eitner, 2016; De Lange et al., 2010). Two maps for the ecological vulnerability, including potential biodiversity and ecological integrity were constructed using two surrogates: land use and protection status of natural areas. These aim at developing the PEB, to evaluate the ecological vulnerability for the NEA. This does not pretend to be an exhaustive assessment because, as discussed before, other parameters could be integrated to the model. Sierra et al. (2002) have already provided a vulnerability assessment at a coarse scale of 1:500,000 for continental Ecuador. But this is a first important step to assess ecological vulnerability at finer scales ranging from 1:25,000 to 1:250,000 in the NEA, notably to couple it with potential oil hazards but also multiple hazards.

V.4. Groundwater vulnerability as a case study for risk assessment

Risk is a complex term that usually considers the combination of hazards and the components of vulnerability. The susceptibility of the system, its resilience and exposure should be included in the evaluation process. In this study, there is no specific analysis of exposure, but in a geographical and spatial sense, the exposure is expressed with the overlaid maps of vulnerability and hazard. The groundwater example considered in this thesis reveals this aspect. The expression R = f [v, h] does not explicitly integrate the variable for exposure, but the use of a product to couple v and h implies that R is nil when v and/or h are equal to zero; then this approach allows to implicitly consider exposure. In a non-spatially explicit evaluation, the exposure should be evaluated separately.

In other words, if a geographical area is found to overlap only one of the components of risk, there is no risk, e.g., vulnerable areas overlapping no hazard. In practice, the opposite is not occurring because the method always states that an area is vulnerable, even if it is considered as low vulnerability (Lahr and Kooistra, 2010). For instance, the DRASTIC model designates low vulnerable zones with values higher than zero, i.e., in this study a minimum value of 4 of 230. Lahr and Kooistra, (2010), establish that exposure in this case refers to the environmental compartments (landscape scale, using biophysical parameters), but not to the organisms within (smaller and higher detailed scale, ecotoxicological domain). The level to which an organism is affected by pollution is more aligned to an ecotoxicological risk assessment, yet if the compartment is not contaminated the organism included within the compartment would neither be. Although the coupled quantitative and overlay index methodology seems sound, the relative degree of subjectivity of these methods is a drawback because it depends on the researcher's expertise and knowledge of the specific area and its ecological functioning (Busico et al., 2017; Gogu and Dassargues, 2000; Kazakis and Voudouris, 2015). In spite of this drawback, it has been widely used in several studies focusing on natural or technological hazards.

As established in the introduction, intrinsic vulnerability is considered when the properties of the system are modelled and weighted, whereas specific vulnerability is considered when the asset under evaluation is exposed to a specific contamination. The DRASTIC depicts intrinsic vulnerability in the sense that the asset is supposed to be equally affected by a potential hazard at each pixel, considering a pollutant whose weight has been defined to assist planners and decision makers in the task of evaluating vulnerability of areas to groundwater contamination (Aller et al., 1987). It has been widely used in nitrate contamination from fertilizers in karstic aquifers (Andreo et al., 2006; Mimi et al., 2012), and integrated land uses including oil activities (Al-Adamat et al., 2003; Babiker et al., 2005). This study is probably the first to implement volume dimensions and site specific pollutants for unlined oil pits in an argillaceous soil media. The sensitivity analysis using the site specific data provided alternative weights that could be used in the NEA. Conversely, some studies have assessed specific vulnerability, adding a parameter for potential anthropic related pollution, such as land use map with wide range of anthropic activities, i.e., *DRASTIC + LU or PEB + oil related pollution.* Likewise, risk assessment (using intrinsic vulnerability and coupling it with actual hazard by a multiplying factor: *DRASTIC x unlined oil pits hazards* or *PEB x oil related pollution* uses the spatial locations of actual hazards, which is what has been emphasized in Chapter 4. In this chapter, the DRASTIC model was defined to measure the intrinsic vulnerability of a system to pollution, given that there is no specific hazard overlaid with actual geographical location of unlined pits.

V.5. Towards integrated ERA

V.5.1. Further assessment of environmental assets

As stated in Chapter III, PEB could also be evaluated via other surrogates, such as habitats and ecosystems, when the quality of data is reliable enough. Prior to the selection of the assets to be chosen in the framework of this thesis, several environmental and social assets were analyzed in terms of feasibility towards integration in spatial risk assessment. It was considered that spatial information available via Ecuadorian institutions did not have enough detail or that heterogeneity was considerably significant, e.g. ecosystem typology, forest fragility. Figure 29 indicates several potential combinations to be evaluated using the material investigated within this thesis. The several combinations for risk assessment are notorious and need to be addressed in a larger context and research programs that should be set up in the near future.

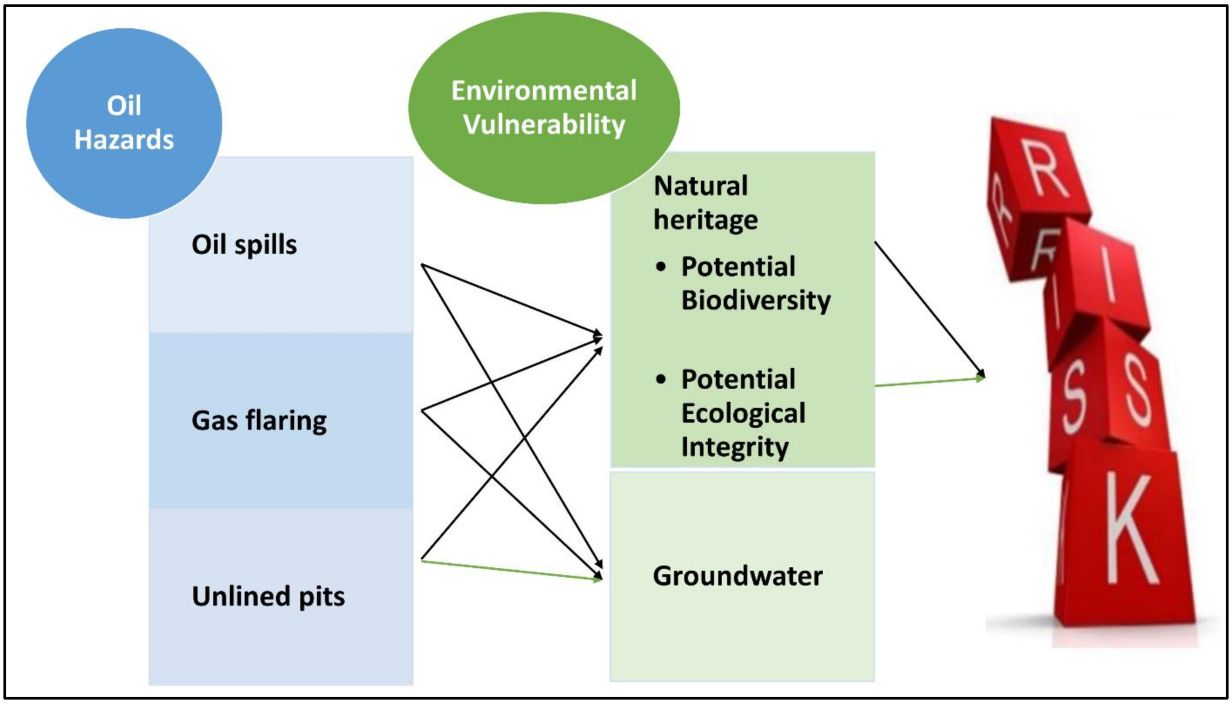


Figure 29. Possible six combinations of risk resulting from the evaluations in this thesis. The green line illustrates the example considered in Chapter IV. The blacked lines represent the combination of spatial risk assessments potentially apprehensible in future studies from the spatial data compiled within this thesis.

Other variables for ecological vulnerability or more general environmental vulnerability could be considered in future studies such as soil fertility (Bahr et al., 2015) that has been already evaluated in zones nearby the NEA (Ochoa-Cueva et al., 2015), namely to evaluate the potential of soils to be subject to erosion and leachate of nutrients in case of disturbance (Dwivedi, 2001; Mainville et al., 2006; Vallejo et al., 2015). Standing biomass could also be a surrogate of ecological vulnerability (Moser et al., 2007). Furthermore, these aforementioned variables could be used to evaluate the environmental resilience of the system.

V.5.2. Selection of alternative assets for vulnerability evaluation

Alternative environmental assets may be selected according to availability of data, feasibility, and technical capacity inherent to the GADs at different administrative levels. Availability of data and technical capacity are self-explained constraints not only inherent to the budget provided to a GAD or a public and para-public institution, but also to the capacity of institutions to optimize the scarce resources and annual budget and objectives establishment in the Land use plans (LU-Plan). Land use plans are developed for a determined time period. They are also related to GADs' technical capacity, denoting their organization skills and their ability to select, upon relevant criteria, the main objectives to target within this context. Several studies have made use of multi-criteria decision-making (He et al., 2018; Malczewski, 2006) or analytical hierarchisation process (Roy and Blaschke, 2015; Thirumalaivasan et al., 2003; Tirkey et al., 2013). These selected data that need to be spatially explicit, but could be derived also from non-spatially explicit databases, e.g., historical associated gas, that could be input to a GIS platform. Some GIS databases have already been explored within this thesis, and are promising for certain environmental assets, i.e., net primary production, soil fertility or biomass, but also for social assets, i.e., population density, public health. While several options of main assets can be considered for a specific site, biodiversity and ecological integrity are unique assets to the NEA, from an environmental perspective and due to their global significance (Bass et al., 2010; Larrea and Warnars, 2009; Mena et al., 2006; Sovacool and Scarpaci, 2016). Therefore, their inclusion within this thesis seemed pertinent. Likewise, groundwater is prone to be contaminated by leachates from unlined oil pits, but its qualification as a representative asset to investigate can generate debates. Despite these arguments, several studies indicate that its importance rises from uncertainty and concerns of its potential pollution risk because it remains a drinkable source for numerous inhabitants of the NEA

(Barraza et al., 2018; Kimerling, 1990, 2015; San Sebastián et al., 2001; Wernersson, 2004). Only 28% of inhabitants in rural area and 72% in urban area have access to potable water (via the drinking water supply network) in the province of Orellana (Province GAD Orellana, 2011).

Alternative options might be the selection of assets, based on methodologies that aim to support decision and consensus making, i.e., multi-criteria analytic hierarchical process, (Roy and Blaschke, 2015) or Delphi methods (Pacheco et al., 2015). These methodological approaches aim to ease decision making about the vulnerabilities that should be given priority, in instances when time and economic resources are scarce. A third option would be to select the assets, based on the priorities for threats and vulnerabilities, established within the land use plans at the province level, which guide the current policies and steps towards a desired future of land use. Otherwise, the decision making process, whether it is participatory approaches, multi-criteria, consensus reaching or hierarchisation to define the vulnerabilities and threats, during the development of LU-Plans should integrate these methods in the first place.

V.5.3. Insights for land use planning implications: towards adaptive capacity management

In the context of vulnerability, the systems' adaptive capacity can be tackled by providing insights for policy interventions to reduce vulnerability (Adger, 2006). The coping ability can be apprehended successfully and practically from the land use planning perspective and it is site specific (Smit and Wandel, 2006). The NEA region comprises the provinces of Sucumbíos and Orellana. These two provincial GADs elaborate their development and land use plans for a defined time period, i.e., Sucumbíos (2011-2020) and Orellana (2015-2019) respectively. The LU-Plans have established the main perceived anthropogenic and natural threats to socio-environmental assets. Sucumbíos and Orellana's LU-Plans indicate that hazards from anthropogenic sources are deforestation, oil production, wild species trafficking, agroindustry, invasive species, disorganized tourism, road opening, hunting and fishing, boundaries, natural heritage borders and frontier trespassing (buffer zones), and other threats to environmental quality (Province GAD Orellana, 2011; Province GAD Sucumbios, 2013). The human basic needs and services are currently improving in the NEA. Furthermore, LU-Plans do not consider vulnerability mapping for groundwater or biodiversity. These plans recognize that unclear geographical limits and colonization are regarded as a threat for natural heritage sites. The problem is exacerbated by the fact that the main source of revenue is mining and oil, reaching up to 94% of the provincial gross domestic product (Province GAD Orellana, 2011). Hence, these activities provide the main income source and might be the main threat at the same time. Diversifying the economy is suggested to be a potential relief to these threats (Berkes, 2007; Ellis, 2008). Social-environmental vulnerabilities are state driven by the fact that still 17% of lands are untitled (Province GAD Orellana, 2011; Province GAD Sucumbios, 2013). Conversely, voluntarily isolated and minority populations' territories in the NEA abbreviated as ZITT (intangible zones) are threatened due to economic interests of extractive industries upon the natural resources that underlie their protected territories.

Land use planning and management foster the proper socio-economic development of a region. The potential economic benefits of using natural resources, i.e., associated gas, commonly wasted during oil production could boost this development, while avoiding atmospheric emissions (World Bank Group, 2018). The benefits would extend to avoiding pollution of agriculture soils and drinkable water (Barraza et al., 2018; Wernersson, 2004). Economic losses from atmospheric emissions and spills towards soil and waters resulted in a total of US$1,523.7 (air) + US$ 724 (spills) million dollars respectively (totaled US$2,247.7) in the NEA; atmospheric emissions consider monetary value for not using APG, excluding health and environment costs (US. Department of Energy, 2017). On the other hand, spill emission costs include economic loss from not selling which is site specific to the NEA, added to remediation and preventive measures: fishery, farming, tourism related costs, damage to property and environmental damage (Kontovas et al., 2010). In this study, economic losses from pits or produced waters were not evaluated because there were no means to quantify the releases and apparently no other studies have evaluated economic costs from these sources.

The attempt of this study to evaluate monetarily the damage caused by these emissions is not exhaustive, but enables some discussion for insights about the perception of stakeholders: oil and gas operators, and experts. For instance, the hazardous emissions that were analyzed in the introduction, with the rapid online survey, can be compared. The surveyed stakeholders provide responses that denoted differences in the perception of the importance of the weights to the overall impact from the three main hazardous sources. Likewise, and more important is the possibility, at least partial, to compare these responses to the economic loss estimated within this thesis (for oil spills and flaring gas). For instance, atmospheric emissions weights (3.40), according to the global surveyed stakeholders, is lower than spills (3.60). However, if stakeholder's activity sectors are compared separately, there are some nuances on their perceptions that, if compared to the potential economic loss estimated within this research, can provide some insights to define which of the hazard sources weighs higher on the overall impact to the total environment. For instance, oil operators (airborne emissions weight = 3.0; oil spills weight = 2.7) and public institutions (airborne emissions weight = 4.6; oil spills weight = 2.9) perceived weights that were proportional to economic costs estimated. Conversely, academia sector (airborne emissions weight = 3.3; oil spills weight = 3.7) and non-governmental organizations (airborne emissions weight = 2.5; oil spills weight = 4.0) considered the oil spills to be more important than gas flaring. As previously established, economic losses from atmospheric emissions impacts doubled the economic loss of oil spills related impacts. Evidently, there is a difference in perceptions, indicating a lack of consensus, but also suggesting that these stakeholders might be better aware of the local impacts. However, this study was unable to determine the causes, and anyhow this was out of its scope.

Another aspect for land use planning is the ecological integrity and biodiversity vulnerability assessment (PEB index). Several recommendations, connected to the historical process of the NEA, could be drawn from the vulnerability map provided:

- Land uses with lowest vulnerability scores in LUs class: agriculture intensification methods should be implemented to avoid pressure in higher vulnerable areas. These may be zones where extensive agriculture, road density and population growth have impacted the landscape. High rates of forest logging during colonization process dominated (before T3 lapse, from 1967 to 2000); Agrarian reforms and government incentives to claim forested lands were established. Road network was publicly accessible in most of these areas.

- Land uses with highest vulnerability scores: protection and conservation strategies should be increased/maintained, notably towards primary forest within natural heritage sites. These areas may correspond to the protected areas having conserved the biodiversity in the long-term, where road accessibility is restricted to certain activities, such as oil development, but somehow precluding human settling, except for the rural communities, notably indigenous (Woaorani tribes) and the ZITT where non contacted groups inhabit.

- Medium vulnerability areas are suggested to be areas where ecological contiguity and connectivity should be enhanced and maintained, e.g. through the implementation of agro ecological systems, partially-closed cattle barn, electrified fences. Certain areas correspond to the non-protected primary forests and other areas belong to the communal forested lands within and out of the Socio-Bosque Program. The road accessibility is not homogeneous in these zones, having different road accessibility types throughout the NEA.

These should be combined with the neighboring pixels as well, since a pixel with a high vulnerability score, (PEB =10) but surrounded by a relevant number of pixels (*n = 1, 2…8)* is going to be influenced towards the neighboring average vulnerability encountered. This same situation can be compared to the probability of land use transitions used to predict land use change. This condition notably impacts borders of protected areas or forested patches, and the broader the size the more neighbor pixels there will be with similar PEB scores, thus their PEB index will most likely remain unchanged for longer periods.

V.6. Perspectives

V.6.1. Towards incorporating components for overall vulnerability: environmental resilience

As mentioned in the introduction, vulnerability has three components: exposure, susceptibility and resilience or adaptive capacity. In this thesis, the resilience component has not been specifically addressed, yet some insights can be procured for future research. First, definition of resilience was provided by Holling (1973). He coined the term resilience in the ecological systems context, expressing the capacity of a system to withstand the changes and still persist (a concept now contested with resistance, considered out of the scope of this study). From there, several authors have rebuilt the concept in several social and ecological frameworks. In this study, resilience would indicate (if separately assessed) the capacity of response of a system to regain a functional state after a disturbance (Becerra et al., 2016; Walker et al., 2002). It might be pertinent to mention that some studies have considered biodiversity alone as a potential metric to assess the possible links with resilience of the system (Oliver et al., 2015); thus the importance to assess vulnerability using the PEB index. The ecological resilience in the NEA can be further addressed using some of the following assets: soil fertility, considering the relation to critical limits of key soil properties and processes, restoration techniques to enhance adaptive capacity, i.e., appropriate management (Lal, 1997). In addition, other parameters could be included, such as standing biomass or net primary production, which can have variable outcomes in the resilience calculation depending on the influential factors of the system considered.

Including thresholds to which the system changes to an alternate state can complete the evaluation of vulnerability (Resilience Alliance and Santa Fe Institute, 2018). These evaluations can serve as a progress towards more complete resilience assessments, however, hindrances from incomplete databases showed that ongoing research is needed (Resilience Alliance and Santa Fe Institute, 2018). In the NEA, data is sufficiently available to investigate the net primary production threshold. Following (Vogt et al., 2016), the combination between precipitation and soil types can provide the quantity needed to reduce vulnerability from stressors in the long-term. In Chapter III, the problem of vulnerability can also be understood from highly biodiverse ecosystems whose ecological integrity has been reduced in a way that might not be seen from space, such as the reduction of genetic pools of fauna and flora in fragmented forested areas (Foster et al., 2013; Silbert, 2002). Specifically, forest connectivity management is foreseen to be implemented in order to reconnect habitats from east to west areas in the NEA, whose PEB values and scores are highest.

V.6.2. Towards incorporating components for overall vulnerability: Societal assets

Societal assets are the complementary side of a complete risk assessment in a socio-ecological systems (SES) framework. Barraza et al. (2018) revealed this importance after addressing the actual pollution of metalloids and oil activities related pollutants in the NEA. Social vulnerabilities might be even more difficult to measure because of the complexity of human behavior (Jörn Birkmann and Wisner, 2006). As such, this aspect should be developed in further detailed studies that can be later coupled with environmental vulnerabilities into a socio-environmental risk assessment within a SES framework. Data availability, either spatially explicit or historical databases that can be input to GIS platforms, have already been explored and proofed promising for certain assets, e.g., population density (can be apprehended in a higher detail than administrative levels[27]; for some insights for future studies see Fig.S.1A,1B), poverty incidence, unsatisfied basics needs (Larrea, 2006; SENPLADES, 2013), or indigenous and non-indigenous

[27] Notably, at the parish level that has readily available spatial information.

minority groups. In Ecuador, the Integrated System of Social Indicators (SIISE, 2016) considers the illiteracy, chronic malnutrition, poverty incidence, infant mortality risk and ethnicity (minorities) as parameters for a vulnerability index. Alternatively, assets could be selected based on methodologies that aim to support decision and consensus making, discussed in section V.6.4. As demonstrated in chapter IV, risk assessments could also include social vulnerabilities using the broad spectrum of human activities. Some of the social variables can be easily integrated, i.e., linking certain human behavior to land use or road networks.

V.6.3. Transport models and validation

The spatial inventories of oil-related hazards could be used as input to model contaminant dispersion and trajectory, or to improve risk assessments when associated with vulnerability maps. For instance, in the case study about groundwater vulnerability and risk, the evaluation of total petroleum hydrocarbon transfers and remaining time in the medium can improve the understanding of the hazard and thus the specific vulnerability to hydrocarbon contaminants. Using several factors for describing the physico-chemistry properties and the dynamics of contaminants, i.e., cross-sectional area, partition fractions, biodegradation, sedimentation or chemical complexion, a multi-phase flow approach using in situ measurements could benefit from the use of specialized software, e.g., Comsol, Modflow, Hydrus-1D, etc. (Ngo et al., 2014) and of integrating stochastic techniques, i.e., Markov chain models (Weigand et al., 2001). It represents an advancement towards finer scales and spatial resolutions and, thus improvement for simulating the contamination plume. Site specific scale models for unlined oil pits in their direct downstream area may be applied to evaluate probabilities into a simpler regional scale model using descriptive data obtained within this study. Several potential sampling sites could be delineated for further research to validate the DRASTIC model, and these zones could be selected using the geographical location of pits and the risk levels defined in chapter IV, to measure the actual contamination at different thresholds distances (Ribeiro et al., 2017). Some sites could be randomly selected in areas presenting different levels of risk, and subsequent in situ sampling and pollutant analyses could help validate the maps produced in the frame of this thesis. Local decision-makers and spatial planners might encourage further investigations in proposed high risk zones according to the most likely scenarios. Land use planning may improve if integrating risk and vulnerability assessments, for instance to plan for potential relocation of certain agricultural activities, so they do not overlap these high risk zones, or to adapt the capacities of GADs to provide potable drinking water to certain areas that are in much need. In areas where budget constraints cannot relieve these problems, risk communication to colonists and indigenous groups for improving public health will be necessary. A perceived major problem to deal with, is the accessibility for in situ measurement due to logistic, administrative, and other field constraints. From the evaluation of oil spill and airborne emissions, it is suggested that oil operators have disclosed data, yet it seems that the efforts in this direction could increase. Collaboration of oil operators, data exchange, and best practices are required to foster practices improvement aiming to progress implementation of monitoring and management systems to reduce risk.

VI. - Conclusion

This thesis presented a spatial risk assessment of oil activities at a regional scale in the Northeastern Ecuadorian Amazon. The two components of risk, hazard and vulnerability, were assessed during this work. For illustrative purposes but not exhaustively, this study had to address main hazards and key vulnerabilities. Hazardous emissions, such as black carbon released by gas flaring, or oil spills, provide a general impact and are related to the spatial distribution of oil activities. For instance, since black carbon is simultaneously a climate change driver and an important fraction of airborne particulate matter, it seemed pertinent to integrate it. Likewise, oil spills, in general, contain several heavy metals and hydrocarbons impacting soil and waters. Unlined oil pits related discharges were more difficult to address in the same way, but their integration as an example for demonstration of risk assessment in chapter IV was a first step to future in situ evaluations to complete knowledge gaps about the behavior of pollutants and to validate results presented in this study. In this final chapter, a regional scaled plume was derived. The same approach could be implemented for oil spills, but transport models for airborne emissions remain completely to be addressed. It should be noted that the fate of contaminants in the atmosphere is strongly influenced by wind and other meteorological variables, which complicates the modeling of contaminant transfers in this compartment.

Several new scientific approaches have been introduced in this thesis, i.e., the validation procedure to provide accuracy assessment to spatial prediction for harmonized maps for oil spills, hazard assessment of unlined oil pits using volumetric dimensions and medium composition (relative proportion of soil and hydrocarbons). Nonetheless, several limitations were acknowledged and gaps should be narrowed in future studies. These gaps in knowledge are often due to data heterogeneity, are spatial scale dependent, and related to unsystematic data collection and access to data. All along of this thesis data quality has been attempted to be assessed. In spite of the acceptability of data accessibility and quality for risk assessment at regional scales, it could significantly be improved. Nonetheless, this thesis could only provide insights upon data quality, yet inconclusive results indicate that even though publicly available and governmental institutional data are available for research purposes, the acquisition of specific data is required and may benefit from independent research partnerships.

On the other hand, vulnerability assessment, which is the other component of risk, can generate more debate about the choice of environmental assets, because there are numerous assets that could be considered. In the NEA, the choice of natural heritage seemed pertinent, given the global significance of this region regarding biodiversity and ecological processes. Nonetheless, groundwater is a more debatable choice, because there is uncertainty about the real relevance of groundwater to the ecological processes, biodiversity and well-being of the populations living in the NEA. The poor coverage of potable water system in rural areas and the potential contamination from unlined oil pits (very important historical hazard source from oil activities) can probably pinpoint the relevance to assess this particular asset.

This study pointed out at human settlements and surrounding natural heritage areas that might be highly impacted by oil spills and airborne emissions, such as Joya de los Sachas, Shushufindi and Nueva Loja and other potentially impacted areas revealed from the harmonized map, such as El Coca, Tarapoa, Dícaro, and Yuturi. In addition, Dayuma and Pacayacu, apparently, may be settlements that are particularly highly impacted by airborne emissions. Tarapoa, Dícaro and Yuturi are settlements at highest risk of groundwater contamination, potentially, and, except for El Coca, they are located within protected areas, where more indigenous communities may live in closer relationship with the environment. Human activities within these areas need higher environmental standards or should be avoided, because they are highly vulnerable. Safety and security monitoring should be enhanced and long-term standing biodiversity conserved. The threat posed to these important ecological areas by natural resources, i.e., mining and crude oil reserves beneath them, remains high. These resources represent an important economic revenue for Ecuador, hence their exploitation is expected to continue.

VI. - Conclusion en français

Cette thèse a présenté une évaluation spatiale des risques liés aux activités pétrolières à l'échelle régionale dans le nord-est de l'Amazonie équatorienne. Les deux composantes du risque, l'aléa et la vulnérabilité, ont été évaluées au cours de ce travail. À des fins d'illustration mais de manière non exhaustive, cette étude devait aborder les principaux aléas et les principales vulnérabilités. Les rejets dangereux, tels que le noir de carbone issu des torchères et les déversements d'hydrocarbures sont structurés spatialement en fonction de la localisation des activités pétrolières. Le noir de carbone étant à la fois un facteur de changement climatique et une fraction importante des particules fines atmosphériques, il a semblé pertinent de l'intégrer. De même, les déversements d'hydrocarbures contiennent divers métaux lourds et des hydrocarbures qui ont un impact sur les sols et les eaux. Il faut noter que les rejets liés aux fosses de stockage de résidus pétroliers étaient plus difficiles à traiter de la même manière, mais leur intégration comme cas d'application de l'évaluation des risques au chapitre IV était une première étape pour les futures évaluations *in situ* qui permettront de combler les lacunes dans nos connaissances sur le comportement des polluants et valider les résultats présentés dans cette étude. Dans ce dernier chapitre, un panache d'hydrocarbures a été dérivé à l'échelle régionale. La même approche pourrait être appliquée aux déversements accidentels d'hydrocarbures. En revanche, les modèles de transport pour les émissions atmosphériques restent complètement à traiter. Il faut d'ailleurs noter que le devenir des contaminants dans l'atmosphère est fortement influencé par le vent et par d'autres variables météorologiques, ce qui vient compliquer la modélisation des transferts de contaminants dans ce compartiment.

Plusieurs nouvelles approches scientifiques ont été introduites dans cette thèse, à savoir, la procédure de validation pour fournir une évaluation de la précision des cartes harmonisées pour les déversements accidentels de pétrole brut, l'évaluation des risques pour les fosses de stockage non revêtues, en utilisant leurs dimensions volumétriques et la composition du mélange (proportion relative de sol et d'hydrocarbures). Néanmoins, plusieurs limitations ont été identifiées et les imprécisions qui en résultent devraient être réduites dans les études futures. Ces lacunes dans les connaissances sont souvent dues à l'hétérogénéité des données, qui varie spatialement, à la collecte de données non systématique et à l'accès aux données existantes. Tout au long de cette thèse, la qualité des données a été évaluée. Malgré l'accessibilité des données et leur qualité acceptable pour l'évaluation des risques à l'échelle régionale, les résultats pourraient être considérablement améliorés. Néanmoins, cette thèse n’a pu fournir qu’un premier aperçu de la qualité des données, et les résultats sur ce point ne sont donc pas complètement conclusifs. Quoi qu’il en soit, même si des données institutionnelles et des données publiquement accessibles sont disponibles à des fins de recherche, l’acquisition de données spécifiques dans le cadre de projets de recherche indépendants permettra d’avancer sur la problématique traitée dans cette thèse.

D'autre part, l'évaluation de la vulnérabilité, qui est l'autre composante du risque, peut susciter un débat quant aux enjeux environnementaux sélectionnés pour cette étude, car de nombreux enjeux pourraient être pris en compte. Dans l'AEN, le choix du patrimoine naturel semblait pertinent, compte tenu de l'importance mondiale de cette région pour la biodiversité et les processus écologiques. Néanmoins, les eaux souterraines constituent un choix plus discutable, car l’importance réelle de ces eaux pour les processus écologiques, la biodiversité et le bien-être des populations vivant dans la NEA est incertaine. La faible couverture du réseau d'eau potable dans les zones rurales et la contamination potentielle par des fosses de stockage non revêtues (source historique de risque important liée aux activités pétrolières) justifie probablement la pertinence d'évaluer cet enjeu en particulier.

Cette étude a mis en évidence les peuplements humains et les zones du patrimoine naturel environnant susceptibles d'être fortement affectés par les déversements d’hydrocarbures et les émissions atmosphériques, telles que Joya de los Sachas, Shushufindi et Nueva Loja, ainsi que d'autres zones potentiellement impactées révélées par la carte harmonisée, telles que El Coca, Tarapoa, Dícaro et Yuturi. De plus, Dayuma et Pacayacu pourraient être des agglomérations particulièrement touchées par les émissions atmosphériques. Tarapoa, Dícaro et Yuturi sont potentiellement les zones les plus exposées au risque de contamination des eaux souterraines et, à l'exception d'El Coca, elles sont situées dans des zones protégées, où davantage de communautés autochtones pourraient vivre en relation plus étroite avec l'environnement. Les activités humaines dans ces zones nécessitent des normes environnementales plus strictes ou doivent être évitées car ces zones sont donc extrêmement vulnérables. La surveillance de la sûreté et de la sécurité devrait être améliorée et la biodiversité préservée. La menace qui pèse sur ces zones écologiques importantes en raison des ressources naturelles, en l’occurrence des réserves minières et de pétrole brut dans leur sous-sol, reste élevé. Ces ressources représentent un revenu économique important pour l’Équateur et, par conséquent, leur exploitation devrait se poursuivre.

VII. Bibliography

Abdel-Shafy, H.I., Mansour, M.S.M., 2016. A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. Egypt. J. Pet. 25, 107–123. https://doi.org/10.1016/J.EJPE.2015.03.011

Acosta, A., 2006. Breve historia económica del Ecuador, 3rd ed. Quito, Ecuador.

Adger, W.N., 2006. Vulnerability. Glob. Environ. Chang. 16, 268–281. https://doi.org/10.1016/j.gloenvcha.2006.02.006

Al-Adamat, R.A.N., Foster, I.D.L., Baban, S.M.J., 2003. Groundwater vulnerability and risk mapping for the Basaltic aquifer of the Azraq basin of Jordan using GIS, Remote sensing and DRASTIC. Appl. Geogr. 23, 303–324. https://doi.org/10.1016/j.apgeog.2003.08.007

Aller, L., Lehr, J.H., Petty, R., 1987. DRASTIC: A standardized Sytem to Evaluate Ground Water Pollution Using Hydrogeologic Settings.

Amazon Defense Front, 2008. Breve Historia de las Operaciones de Texaco en el Oriente Ecuatoriano 1964 -1990. Quito, Ecuador.

Ambiente, M. de, 2018. Sistema Nacional de Áreas Protegidas Del Ecuador [WWW Document]. online site. URL http://areasprotegidas.ambiente.gob.ec/en/todas-areas-protegidas-por-region?t=A (accessed 9.21.18).

Andreo, B., Goldscheider, N., Vadillo, I., Vías, J.M., Neukum, C., Sinreich, M., Jiménez, P., Brechenmacher, J., Carrasco, F., Hötzl, H., Perles, M.J., Zwahlen, F., 2006. Karst groundwater protection: First application of a Pan-European Approach to vulnerability, hazard and risk mapping in the Sierra de Líbar (Southern Spain). Sci. Total Environ. 357, 54–73. https://doi.org/10.1016/j.scitotenv.2005.05.019

Anejionu, Blackburn, G.A., Whyatt, J.D., 2015a. Detecting gas flares and estimating flaring volumes at individual flow stations using MODIS data. Remote Sens. Environ. 158, 81–94. https://doi.org/10.1016/j.rse.2014.11.018

Anejionu, Whyatt, J.D., Blackburn, G.A., Price, C.S., 2015b. Contributions of gas flaring to a global air pollution hotspot: Spatial and temporal variations, impacts and alleviation. Atmos. Environ. 118, 184–193. https://doi.org/10.1016/j.atmosenv.2015.08.006

Anenberg, S.C., Schwartz, J., Shindell, D., Amann, M., Faluvegi, G., Klimont, Z., Janssens-Maenhout, G., Pozzoli, L., Van Dingenen, R., Vignati, E., Emberson, L., Muller, N.Z., West, J.J., Williams, M., Demkine, V., Hicks, W.K., Kuylenstierna, J., Raes, F., Ramanathan, V., 2012. Global air quality and health co-benefits of mitigating near-term climate change through methane and black carbon emission controls. Environ. Health Perspect. 120, 831–9. https://doi.org/10.1289/ehp.1104301

ARCH, 2018. Agencia de Regulación y Control Hidrocarburífero [WWW Document]. URL http://www.controlhidrocarburos.gob.ec/indicadores/ (accessed 11.20.18).

Arellano, P., Tansey, K., Balzter, H., Boyd, D.S., 2015. Detecting the effects of hydrocarbon pollution in the Amazon forest using hyperspectral satellite images. Environ. Pollut. 205, 225–239. https://doi.org/10.1016/j.envpol.2015.05.041

Arellano, P., Tansey, K., Balzter, H., Tellkamp, M., 2017. Plant Family-Specific Impacts of Petroleum Pollution on Biodiversity and Leaf Chlorophyll Content in the Amazon Rainforest of Ecuador 1–18. https://doi.org/10.1371/journal.pone.0169867

Aretano, R., Semeraro, T., Petrosillo, I., De Marco, A., Pasimeni, M.R., Zurlini, G., 2015. Mapping ecological vulnerability to fire for effective conservation management of natural protected areas. Ecol. Modell. 295, 163–175. https://doi.org/10.1016/J.ECOLMODEL.2014.09.017

Argo, J., 2001. Unhealthy effects of upstream oil and gas flaring : a report prepared for Save our Seas and Shores (SOSS) for presentation before the public review commission into effects of potential oil and gas exploration, drilling activities [WWW Document]. URL https://www.osti.gov/etdeweb/biblio/20291224 (accessed 10.27.18).

ATSDR, 2019. Barium Compounds [WWW Document]. Agency Toxic Subst. Dis. Regist. URL https://www.atsdr.cdc.gov/toxprofiles/tp24-c8.pdf (accessed 2.5.19).

ATSDR, 2011a. ATSDR-Total Petroleum Hydrocarbons (TPH) [WWW Document]. Agency Toxic Subst. Dis. Regist. URL https://www.atsdr.cdc.gov/toxfaqs/tf.asp?id=423&tid=75 (accessed 10.24.18).

ATSDR, 2011b. ATSDR - Toxic Substances - Polycyclic Aromatic Hydrocarbons (PAHs) [WWW Document]. Agency Toxic Subst. Dis. Regist. https://doi.org/83-32-9, 120-12-7

Babiker, I.S., Mohamed, M.A.A., Hiyama, T., Kato, K., 2005. A GIS-based DRASTIC model for assessing aquifer vulnerability in Kakamigahara Heights, Gifu Prefecture, central Japan. Sci. Total Environ. 345, 127–140. https://doi.org/10.1016/j.scitotenv.2004.11.005

Bahr, E., Chamba-Zaragocin, D., Fierro-Jaramillo, N., Witt, A., Makeschin, F., 2015. Modeling of soil nutrient balances, flows and stocks revealed effects of management on soil fertility in south Ecuadorian smallholder farming systems. Nutr. Cycl. Agroecosystems 101, 55–82. https://doi.org/10.1007/s10705-014-9662-5

Bakke, T., Klungsøyr, J., Sanni, S., 2013. Environmental impacts of produced water and drilling waste discharges from the Norwegian offshore petroleum industry. Mar. Environ. Res. 92, 154–169. https://doi.org/10.1016/j.marenvres.2013.09.012

Barraza, F., Maurice, L., Uzu, G., Becerra, S., López, F., Ochoa-Herrera, V., Ruales, J., Schreck, E., 2018. Distribution, contents and health risk assessment of metal(loid)s in small-scale farms in the Ecuadorian Amazon: An insight into impacts of oil activities. Sci. Total Environ. 622–623, 106–120. https://doi.org/10.1016/j.scitotenv.2017.11.246

Barraza, F., Schreck, E., Lévêque, T., Uzu, G., López, F., Ruales, J., Prunier, J., Marquet, A., Maurice, L., 2017. Cadmium bioaccumulation and gastric bioaccessibility in cacao: A field study in areas impacted by oil activities in Ecuador. Environ. Pollut. 229, 950–963. https://doi.org/10.1016/j.envpol.2017.07.080

Bass, M.S., Finer, M., Jenkins, C.N., Kreft, H., Cisneros-Heredia, D.F., McCracken, S.F., Pitman, N.C. a, English, P.H., Swing, K., Villa, G., Di Fiore, A., Voigt, C.C., Kunz, T.H., 2010. Global conservation significance of Ecuador's Yasuní National Park. PLoS One 5.

https://doi.org/10.1371/journal.pone.0008767

Baynard, C.W., Ellis, J.M., Davis, H., 2013. Roads, petroleum and accessibility: The case of eastern Ecuador. GeoJournal 78, 675–695. https://doi.org/10.1007/s10708-012-9459-5

Bayramin, İ., Basaran, M., Erpul, G., Canga, M.R., 2008. Assessing the effects of land use changes on soil sensitivity to erosion in a highland ecosystem of semi-arid Turkey. Environ. Monit. Assess. 140, 249–265. https://doi.org/10.1007/s10661-007-9864-2

BBC News, 2018. Chevron wins Ecuador rainforest “oil dumping” case - BBC News [WWW Document]. Sept. 08. URL https://www.bbc.co.uk/news/world-latin-america-45455984 (accessed 9.13.18).

Becerra, S., Saqalli, M., Gangneron, F., Dia, A.H., 2016. Everyday vulnerabilities and “social dispositions” in the Malian Sahel, an indication for evaluating future adaptability to water crises? Reg. Environ. Chang. 16, 1253–1265. https://doi.org/10.1007/s10113-015-0845-7

Benkovitz, C.M., Scholtz, M.T., Pacyna, J., Tarrasón, L., Dignon, J., Voldner, E.C., Spiro, P.A., Logan, J.A., Benkovitz, C.M., Scholtz, M.T., Pacyna, J., Tarrasón, L., 1996. Global gridded inventories of anthropogenic emissions of sulfur and nitrogen Global gridded inventories of anthropogenic emissions of sulfur and nitrogen.

Berkes, F., 2007. Understanding uncertainty and reducing vulnerability: Lessons from resilience thinking. Nat. Hazards 41, 283–295. https://doi.org/10.1007/s11069-006-9036-7

Bernard, E., Fenton, M.B., 2002. Species diversity of bats (Mammalia: Chiroptera) in forest fragments, primary forests, and savannas in central Amazonia, Brazil. Can. J. Zool. 80, 1124–1140. https://doi.org/10.1139/z02-094

Beroya-Eitner, M.A., 2016. Ecological vulnerability indicators. Ecol. Indic. 60, 329–334. https://doi.org/10.1016/j.ecolind.2015.07.001

Berry, P.M., Rounsevell, M.D.A., Harrison, P.A., Audsley, E., 2006. Assessing the vulnerability of agricultural land use and species to climate change and the role of policy in facilitating adaptation. Environ. Sci. Policy 9, 189–204. https://doi.org/10.1016/j.envsci.2005.11.004

Bertazzon, S., O’Hara, P.D., Barrett, O., Serra-Sogas, N., 2014. Geospatial analysis of oil discharges observed by the National Aerial Surveillance Program in the Canadian Pacific Ocean. Appl. Geogr. 52, 78–89. https://doi.org/10.1016/J.APGEOG.2014.04.010

Beukema, H., Van Noordwijk, M., 2004. Terrestrial pteridophytes as indicators of a forest-like environment in rubber production systems in the lowlands of Jambi, Sumatra. Agric. Ecosyst. Environ. 104, 63–73. https://doi.org/10.1016/j.agee.2004.01.007

Birkmann, J., 2011. Risk and vulnerability indicators at different scales : Applicability , usefulness and policy implications Risk and vulnerability indicators at different scales : Applicability , usefulness and policy implications 7891. https://doi.org/10.1016/j.envhaz.2007.04.002

Birkmann, J., Wisner, B., 2006. Measuring the un-measurable the challenge of vulnerability. UNU EHS, Bonn, Germany.

Birkmann, J., Wisner, B., 2006. Measuring the un-measurable:The Challenge of Vulnerability, The challenge of vulnerability. Source.

Bissardon, P., Becerra, S., Maurice, L., 2013. Le risque sanitaire lié aux activités pétrolières en Amazonie équatorienne: des alertes aux décisions. Env. Risque Sante 12, 338–344. https://doi.org/doi : 10.1684/ers.2013.0627

Boden, T., Marland, G., Andres, B., 2012. National Carbone Dioxide Emissions from Fossil-fuel, Cement Manufacture and Gas flaring: 1751-2008 [WWW Document]. Carbon Dioxide Inf. Anal. Center. Oak Ridge Natl. Lab. URL http://cdiac.ess-dive.lbl.gov/trends/emis/ecu.html (accessed 4.30.18).

Bohle, H.G., Downing, T.E., Watts, M.J., 1994. Climate change and social vulnerability: Toward a sociology and geography of food insecurity. Glob. Environ. Chang. 4, 37–48. https://doi.org/10.1016/0959-3780(94)90020-5

Bond, T.C., Bergstrom, R.W., 2006. Light Absorption by Carbonaceous Particles: An Investigative Review. Aerosol Sci. Technol. 40, 27–67. https://doi.org/10.1080/02786820500421521

Bond, T.C., Streets, D.G., Yarber, K.F., Nelson, S.M., Woo, J.H., Klimont, Z., 2004. A technology-based global inventory of black and organic carbon emissions from combustion. J. Geophys. Res. D Atmos. 109, 1–43. https://doi.org/10.1029/2003JD003697

Bondur, V.G., 2011. Aerospace methods and technologies for monitoring oil and gas areas and facilities. Izv. Atmos. Ocean. Phys. 47, 1007–1018. https://doi.org/10.1134/S0001433811090039

Bonilla-Bedoya, S., Molina, J.R., Macedo-Pezzopane, J.E., Herrera-Machuca, M.A., 2014. Fragmentation patterns and systematic transitions of the forested landscape in the upper Amazon region, Ecuador 1990-2008. J. For. Res. 25, 301–309. https://doi.org/10.1007/s11676-013-0419-9

Boxall, P.C., Chan, W.H., Mcmillan, M.L., 2005. The impact of oil and natural gas facilities on rural residential property values : a spatial hedonic analysis. Resour. Energy Econ. 27, 248–269. https://doi.org/10.1016/j.reseneeco.2004.11.003

Brinck, K., Fischer, R., Groeneveld, J., Lehmann, S., Dantas De Paula, M., Pütz, S., Sexton, J.O., Song, D., Huth, A., 2017. High resolution analysis of tropical forest fragmentation and its impact on the global carbon cycle. Nat. Commun. 8, 14855. https://doi.org/10.1038/ncomms14855

Bronfman, N.C., López, E., Dorantes, G., 2009. An empirical study for the direct and indirect links between trust in regulatory institutions and acceptability of hazards. Saf. Sci. 47, 686–692. https://doi.org/10.1016/j.ssci.2008.09.006

Buccina, S., Chene, D., Gramlich, J., 2013. Accounting for the environmental impacts of Texaco's operations in Ecuador: Chevron's contingent environmental liability disclosures. Account. Forum 37, 110–123. https://doi.org/10.1016/j.accfor.2013.04.003

Burbano, N., Becerra, S., Pasquel, E., 2015. Introducción a la Hidrogeología del Ecuador. Quito, Ecuador.

Burgherr, P., 2007. In-depth analysis of accidental oil spills from tankers in the context of global spill trends from all sources. J. Hazard. Mater. 140, 245–256. https://doi.org/10.1016/j.jhazmat.2006.07.030

Burgherr, P., Eckle, P., Hirschberg, S., 2012. Comparative assessment of severe accident risks in the coal, oil and natural gas chains. Reliab. Eng. Syst. Saf. 105, 97–103. https://doi.org/10.1016/j.ress.2012.03.020

Burgherr, P., Hirschberg, S., 2014. Comparative risk assessment of severe accidents in the energy sector. Energy Policy 74, S45–S56. https://doi.org/10.1016/j.enpol.2014.01.035

Busico, G., Colombani, N., Cuoco, E., Mastrocicco, M., Sirna, M., Tedesco, D., 2017. Modifying SINTACS method to assess groundwater vulnerability and pollution risk to nitrate, European Water.

Butt, N., Beyer, H.L., Bennett, J.R., Biggs, D., Maggini, R., Mills, M., Renwick, a. R., Seabrook, L.M., Possingham, H.P., 2013. Biodiversity risks from fossil fuel extraction. Science (80-.). 342, 425–426. https://doi.org/10.1126/science.1237261

Cabrera, R., 2008. Extrapolation of sampling results from all sites within the Texaco-Chevron oil concession blocs. Quito.

Castello, L., Mcgrath, D.G., Hess, L.L., Coe, M.T., Lefebvre, P.A., Petry, P., Macedo, M.N., Ren, V.F., Arantes, C.C., 2013. The vulnerability of Amazon freshwater ecosystems. Conserv. Lett. 217–229. https://doi.org/10.1111/conl.12008

Castelo, F.B., 2017. Human exposure assessment related to oil activities in Ecuador : from the air quality monitoring to the study of metallic contaminants transfer in the soil-plant.

Cebrián-Piqueras, M.A., Trinogga, J., Grande, C., Minden, V., Maier, M., Kleyer, M., 2017. Interactions between ecosystem properties and land use clarify spatial strategies to optimize trade-offs between agriculture and species conservation. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 13, 53–66. https://doi.org/10.1080/21513732.2017.1289245

Chang, S., Stone, J., Demes, K., Piscitelli, M., 2014. Consequences of oil spills: a review and framework for informing planning. Ecol. Soc. 19, 26. https://doi.org/10.5751/ES-06406-190226

Chaudhary, A., Verones, F., De Baan, L., Hellweg, S., 2015. Quantifying Land Use Impacts on Biodiversity: Combining Species-Area Models and Vulnerability Indicators. Environ. Sci. Technol. 49, 9987–9995. https://doi.org/10.1021/acs.est.5b02507

Conrad, B., Johnson, M., 2015. Field Measurements Of Black Carbon Emissions From Flaring In Ecuador: Analysis of Results from Field Measurements.

Conrad, B.M., Johnson, M.R., 2017. Field Measurements of Black Carbon Yields from Gas Flaring. Environ. Sci. Technol. 51, 1893–1900. https://doi.org/10.1021/acs.est.6b03690

Cuba, N., Bebbington, A., Rogan, J., Millones, M., 2014. Extractive industries, livelihoods and natural resource competition: Mapping overlapping claims in Peru and Ghana. Appl. Geogr. 54, 250–261. https://doi.org/10.1016/j.apgeog.2014.05.003

Cutter, S.L., 1996. Vulnerability to environmental hazards. Prog. Hum. Geogr. https://doi.org/10.1177/030913259602000407

Darbra, R.M., Palacios, A., Casal, J., 2010. Domino effect in chemical accidents: Main features and accident sequences. J. Hazard. Mater. 183, 565–573. https://doi.org/10.1016/j.jhazmat.2010.07.061

De Lange, H.J., Lahr, J., Van Der Pol, J.J.C., Wessels, Y., Faber, J.H., 2009. Ecological vulnerability in wildlife: An expert judgment and multicriteria analysis tool using ecological traits to assess relative impact of pollutants. Environ. Toxicol. Chem. 28, 2233–2240. https://doi.org/10.1897/08-626.1

De Lange, H.J., Sala, S., Vighi, M., Faber, J.H., 2010. Ecological vulnerability in risk assessment - A review and perspectives. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2009.11.009

De Miranda, R.M., Andrade, M. de F., Fornaro, A., Astolfo, R., de Andre, P.A., Saldiva, P., 2012. Urban air pollution: A representative survey of PM 2.5 mass concentrations in six Brazilian cities. Air Qual. Atmos. Heal. 5, 63–77. https://doi.org/10.1007/s11869-010-0124-1

Dlugokencky, E.J., 2003. Atmospheric methane levels off: Temporary pause or a new steady-state? Geophys. Res. Lett. 30, 3–6. https://doi.org/10.1029/2003GL018126

Dudley, N., 2008. Guidelines for Protected Area Management Categories. System 3, 86. https://doi.org/10.2305/IUCN.CH.2008.PAPS.2.en

Dumbrell, A.J., Hill, J.K., 2005. Impacts of selective logging on canopy and ground assemblages of tropical forest butterflies: Implications for sampling. Biol. Conserv. 125, 123–131. https://doi.org/10.1016/j.biocon.2005.02.016

Dung, E.J., Bombom, L.S., Agusomu, T.D., 2008. The effects of gas flaring on crops in the Niger Delta, Nigeria. GeoJournal 73, 297–305. https://doi.org/10.1007/s10708-008-9207-z

Dwivedi, R.S., 2001. Soil resources mapping: A remote sensing perspective. Remote Sens. Rev. 20, 89–122. https://doi.org/10.1080/02757250109532430

Eggleton, P., Bignell, D.E., Sands, W.A., Waite, B., Wood, T.G., Lawton, J.H., Journal, S., Feb, N., 2016. The Species Richness of Termites (Isoptera) Under Differing Levels of Forest Disturbance in the Published by : Cambridge University Press Stable URL : http://www.jstor.org/stable/2560144 Accessed : 17-03-2016 14 : 14 UTC Your use of the JSTOR archive in. J. Trop. Ecol. 11, 85–98.

Egoh, B., Reyers, B., Rouget, M., Bode, M., Richardson, D.M., 2009. Spatial congruence between biodiversity and ecosystem services in South Africa. Biol. Conserv. 142, 553–562. https://doi.org/10.1016/J.BIOCON.2008.11.009

Egoh, B.N., Reyers, B., Rouget, M., Richardson, D.M., 2011. Identifying priority areas for ecosystem service management in South African grasslands. J. Environ. Manage. 92, 1642–1650. https://doi.org/10.1016/J.JENVMAN.2011.01.019

Egwurugwu, J.N., Nwafor, A., Oluronfemi, O.J., C, I.S., A, A.E., 2013. Impact of Prolonged Exposure to Oil and Gas Flares on Human Renal Functions. Int. Res. J. Med. Sci. Int . Res . J . Med. Sci 1, 2320–7353.

Ellis, F., 2008. The Determinants of Rural Livelihood Diversification in Developing Countries. J. Agric. Econ. 51, 289–302. https://doi.org/10.1111/j.1477-9552.2000.tb01229.x

EPA, 2014. Compilation of Publicly Available Sources of Voluntary Management Practices for Oil and Gas Exploration & Production (E&P) Wastes As They Address Pits, Tanks, and Land Application [WWW Document]. URL https://www.epa.gov/hw/proper-management-oil-and-gas-exploration-and-production-waste

Erickson, R.J., Mount, D.R., Highland, T.L., Russell Hockett, J., Jenson, C.T., 2011. The relative importance of waterborne and dietborne arsenic exposure on survival and growth of juvenile rainbow trout. Aquat. Toxicol. 104, 108–115. https://doi.org/10.1016/j.aquatox.2011.04.003

Estrada, A., Coates-estrada, R., Meritt, D., 1993. Bat species richness and abundance in tropical rain forest habitats at Los Thxtlas , Mexico fragments and in agricultural. Wiley behalf Nord. Soc. Oikos Stable 16, 309–318.

Fabian, P., Kohlpaintner, M., Rollenbeck, R., 2005. Biomass Burning in the Amazon-Fertilizer for the Mountaineous Rain Forest in Ecuador. Env. Sci Pollut Res 12, 290–296. https://doi.org/10.1065/espr2005.07.272

Faria, D., Laps, R.R., Baumgarten, J., Cetra, M., 2006. Bat and bird assemblages from forests and shade cacao plantations in two contrasting landscapes in the Atlantic Forest of southern Bahia, Brazil. Biodivers. Conserv. 15, 587–612. https://doi.org/10.1007/s10531-005-2089-1

Faria, D., Paciencia, M.L.B., Dixo, M., Laps, R.R., Baumgarten, J., 2007. Ferns, frogs, lizards, birds and bats in forest fragments and shade cacao plantations in two contrasting landscapes in the Atlantic forest, Brazil. Biodivers. Conserv. 16, 2335–2357. https://doi.org/10.1007/s10531-007-9189-z

Fearnside, P.M., 2000. Global Warming and Tropical Land-Use Change: Greenhouse Gas Emissions from Biomass Burning, Decomposition and Soils in Forest Conversion, Shifting Cultivation and Secondary Vegetation. Clim. Change 46, 115–158. https://doi.org/10.1023/A:1005569915357

Ferraro, P.J., Hanauer, M.M., Miteva, D.A., Canavire-Bacarreza, G.J., Pattanayak, S.K., Sims, K.R.E., 2013. More strictly protected areas are not necessarily more protective: Evidence from Bolivia, Costa Rica, Indonesia, and Thailand. Environ. Res. Lett. 8. https://doi.org/10.1088/1748-9326/8/2/025011

Finer, M., Jenkins, C.N., Pimm, S.L., Keane, B., Ross, C., 2008. Oil and Gas Projects in the Western Amazon: Threats to Wilderness, Biodiversity, and Indigenous Peoples. PLoS One 3, e2932. https://doi.org/10.1371/journal.pone.0002932

Finer, M., Vijay, V., Ponce, F., Jenkins, C.N., Kahn, T.R., 2009. Ecuador's Yasuní Biosphere Reserve: a brief modern history and conservation challenges. Environ. Res. Lett 4, 34005–15. https://doi.org/10.1088/1748-9326/4/3/034005

Flax, L., Russell, J., David, S., 2002. Community Assessment Vulnerability Tool. Charleston, SC.

Fontaine, G., Narvaez, I., 2007. Yasuní en el siglo XXI: el Estado ecuatoriano y la conservación de la Amazonía, Abya Ayala. ed, FLACSO. Quito, Ecuador.

Foss, B., 2012. Control Engineering Practice Process control in conventional oil and gas fields — Challenges and opportunities. Control Eng. Pract. 20, 1058–1064. https://doi.org/10.1016/j.conengprac.2011.11.009

Foster, S., Hirata, R., Andreo, B., 2013. The aquifer pollution vulnerability concept: aid or impediment in promoting groundwater protection? Hydrogeol. J. 21, 1389–1392. https://doi.org/10.1007/s10040-013-1019-7

Freitas, S.R., Longo, K.M., Silva Dias, M.A.F., Chatfield, R., Dias, P.S., Artaxo, P., Andreae, M.O., Grell, G., Rodrigues, L.F., Fazenda, A., Panetta, J., 2009. The coupled aerosol and tracer transport model to the Brazilian developments on the regional atmospheric modeling system (CATT-BRAMS)-Part 1: Model description and evaluation. Atmos. Chem. Phys. 9, 2843–2861. https://doi.org/10.5194/acp-9-2843-2009

Fried, J.J., Muntzer, P., Zilliox, L., 1979. Groundwater Pollution by Transfer of Oil Hydrocarbons. Groundwater 17, 586–596.

Front of Amazon Defence, 2008. Breve Historia de las Operaciones de Texaco en el Oriente Ecuatoriano 1964 -1990. Quito, Ecuador.

Frynas, J.G., 2012. Corporate Social Responsibility or Government Regulation ? Evidence on. Ecol. Soc. 17, art4. https://doi.org/10.5751/ES-05073-170404

Füssel, H.-M., 2007. Vulnerability: A generally applicable conceptual framework for climate change research. Glob. Environ. Chang. 17, 155–167. https://doi.org/10.1016/J.GLOENVCHA.2006.05.002

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A., Sodhi, N.S., 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecol. Lett. 12, 561–582. https://doi.org/10.1111/j.1461-0248.2009.01294.x

Gardner, T.A., Ribeiro-Júnior, M.A., Barlow, J., Ávila-Pires, T.C.S., Hoogmoed, M.S., Peres, C.A., 2007. The value of primary, secondary, and plantation forests for a neotropical herpetofauna. Conserv. Biol. 21, 775–787. https://doi.org/10.1111/j.1523-1739.2007.00659.x

Getis, A., Ord, J.K., 1992. The Analysis of Spatial Association by Use of Distance Statistics. Geogr. Anal. 24, 189–206. https://doi.org/10.1111/j.1538-4632.1992.tb00261.x

Giwa, S.O., Adama, O.O., Akinyemi, O.O., 2014. Baseline black carbon emissions for gas flaring in the Niger Delta region of Nigeria. J. Nat. Gas Sci. Eng. 20, 373–379. https://doi.org/10.1016/j.jngse.2014.07.026

Gleyze, J.F., 2002. Le Risque. Insititut Geographique National.

Gogu, R.., Dassargues, A., 2000. Current and future trends in groundwater vulnerability assessment using overlay and index methods. Environ. Geol. 39, 549–559. https://doi.org/https://doi.org/10.1007/s002540050466

Gómez, M.A., 2013. The Global Chase : Seeking the Recognition and Enforcement of the Lago Agrio Judgment Outside of Ecuador 1.

Gower, J.C., 1971. A General Coefficient of Similarity and Some of Its Properties. Biometrics 27, 857–871.

Gramsch, E., Le Nir, G., Araya, M., Rubio, M.A., Moreno, F., Oyola, P., 2013. Influence of large changes in public transportation (Transantiago) on the black carbon pollution near streets. Atmos. Environ. 65, 153–163. https://doi.org/10.1016/J.ATMOSENV.2012.10.006

Guerra del Hierro, G.F., 2014. Estimation of the Oil Production Potential of the Field Singue, Oriente Basin, Ecuador. University of Salford.

Guttikunda, S.K., Calori, G., 2013. A GIS based emissions inventory at 1 km × 1 km spatial resolution for air pollution analysis in Delhi, India. Atmos. Environ. 67, 101–111. https://doi.org/10.1016/j.atmosenv.2012.10.040

Haines, A., Kovats, R.S., Campbell-Lendrum, D., Corvalan, C., 2006. MINI-SYMPOSIUM Climate change and human health: Impacts, vulnerability and public health. https://doi.org/10.1016/j.puhe.2006.01.002

Hamza, M.H., Added, A., Rodríguez, R., Abdeljaoued, S., Ben Mammou, A., 2007. A GIS-based DRASTIC vulnerability and net recharge reassessment in an aquifer of a semi-arid region (Metline-Ras Jebel-Raf Raf aquifer, Northern Tunisia). J. Environ. Manage. 84, 12–19. https://doi.org/10.1016/j.jenvman.2006.04.004

Haro-Carrión, X., Lozada, T., Navarrete, H., De Koning, G.H.J., 2009. Conservation of vascular epiphyte diversity in shade cacao plantations in the Chocó region of ecuador. Biotropica 41, 520–529. https://doi.org/10.1111/j.1744-7429.2009.00510.x

Harvey, C.A., Gonzalez, J., Somarriba, E., 2006. Dung beetle and terrestrial mammal diversity in forests, indigenous agroforestry systems and plantain monocultures in Talamanca, Costa Rica. Biodivers. Conserv. 15, 555–585. https://doi.org/10.1007/s10531-005-2088-2

Harvey, C.A., Medina, A., Sánchez, D.M., Vílchez, S., Saenz, J.C., Maes, J.M., Casanoves, F., Sinclair, F.L., Harvey, C.A., Medina, A., Sanchez, D.M., Vilchez, S., Hernandez, B., Saenz, J.C., Maes, J.M., Casanoves, F., Sinclair, F.L., 2016. Patterns of Animal Diversity in Different Forms of Tree Cover in Agricultural Landscapes. Ecol. Appl. 16, 1986–1999.

Hawes, J., Da Silva Motta, C., Overal, W.L., Barlow, J., Gardner, T.A., Peres, C.A., 2009. Diversity and composition of Amazonian moths in primary, secondary and plantation forests. J. Trop. Ecol. 25, 281–300. https://doi.org/10.1017/S0266467409006038

He, L., Shen, J., Zhang, Y., 2018. Ecological vulnerability assessment for ecological conservation and environmental management. J. Environ. Manage. 206, 1115–1125. https://doi.org/10.1016/J.JENVMAN.2017.11.059

Henzen, C., 2008. The Impact of Land Use on Biodiversity in the Framework of Life Cycle Assessment. University of Basel.

Holland, J., Fahrig, L., 2000. Effect of woody borders on insect density and diversity in crop fields: a landscape-scale analysis. Agric. Ecosyst. Environ. 78, 115–122. https://doi.org/10.1016/S0167-8809(99)00123-1

Holland, M.B., de Koning, F., Morales, M., Naughton-Treves, L., Robinson, B.E., Suárez, L., 2014. Complex Tenure and Deforestation: Implications for Conservation Incentives in the Ecuadorian Amazon. World Dev. 55, 21–36. https://doi.org/10.1016/j.worlddev.2013.01.012

Holling, C.S., 1973. Resilience and Stability of Ecological Systems. Annu. Rev. Ecol. Syst. 4, 1–23.

Huang, K., Fu, J.S., 2016. A global gas flaring black carbon emission rate dataset from 1994 to 2012. Sci. Data. https://doi.org/10.1038/sdata.2016.104

Huq, N., Bruns, A., Ribbe, L., 2019. Interactions between freshwater ecosystem services and land cover changes in southern Bangladesh: A perspective from short-term (seasonal) and long-term (1973–2014) scale. Sci. Total Environ. 650, 132–143. https://doi.org/10.1016/j.scitotenv.2018.08.430

Hurtig, A.K., San Sebastián, M., 2002. Geographical differences in cancer incidence in the Amazon basin of Ecuador in relation to residence near oil fields. Int. J. Epidemiol. 31, 1021–1027. https://doi.org/10.1093/ije/31.5.1021

IARC, 2012. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans. [WWW Document]. URL https://monographs.iarc.fr/list-of-classifications-volumes/ (accessed 9.6.18).

IGN, 2018. Inventaire Forestier [WWW Document]. Insitutut l'informatique Geogr. For. URL https://inventaire-forestier.ign.fr/ (accessed 10.18.18).

INAMHI, 2018. Mapas Climáticos [WWW Document]. Natl. Inst. Metereology Hidrol. Quito, Ecuador. URL http://www.serviciometeorologico.gob.ec/mapas-climaticos/ (accessed 9.21.18).

INEC, 2017. Encuestas de Transporte 2007-2016 [WWW Document]. Inst. Nac. Estadísticas y Censos. Quito, Ecuador. URL http://anda.inec.gob.ec/anda/index.php/catalog#_r=1503587423217&collection=&country=&dtype=&from=2007&page=1&ps=&sk=transporte&sort_by=titl&sort_order=&to=2016&topic=&view=s&vk= (accessed 8.24.17).

IPCC, 2013. Fifth Assessment Report - Climate Change 2013, IPCC. https://doi.org/10.1126/science.1061457

IPCC, 2006a. Chapter 4: Fugitive Emissions Guidelines for National Greenhouse Gases Inventories. Geneva, Switzerland.

IPCC, 2006b. Chapter 3: Uncertainties 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Geneva, Switzerland.

IPCC Software, 2013. IPCC - Task Force on National Greenhouse Gas Inventories. IPCC Invent. Softw.

Isichei, A.O., Sanford, W.W., 1976. The Effects of Waste Gas Flares on the Surrounding Vegetation in South-Eastern Nigeria, Source: Journal of Applied Ecology.

Ite, A., Ibok, U., 2013. Gas Flaring and Venting Associated with Petroleum Exploration and Production in the Nigeria's Niger Delta. Am. J. Environ. Prot. 1, 70–77. https://doi.org/10.12691/env-1-4-1

Ite, A., Ibok, U., Ite, M., Petters, S., 2013. Petroleum Exploration and Production: Past and Present Environmental Issues in the Nigeria's Niger Delta. Am. J. Environ. Prot. 1, 78–90. https://doi.org/10.12691/env-1-4-2

IUCN, 2016. Protected Area Categories [WWW Document]. IUCN World Conserv. Congr. 2016. URL https://www.iucn.org/theme/protected-areas/about/protected-areas-categories (accessed 5.21.18).

Jacobson, M.Z., 2001. Strong radiative heating due to the mixing state of black carbon in atmospheric aerosols. Nature 409, 695–697. https://doi.org/10.1038/35055518

Janssen, N. a. H., Hoek, G., Simic-Lawson, M., Fischer, P., van Bree, L., ten Brink, H., Keuken, M., Atkinson, R.W., Anderson, H.R., Brunekreef, B., Cassee, F.R., 2011. Black Carbon as an Additional Indicator of the Adverse Health Effects of Airborne Particles Compared with

PM10 and PM2.5. Environ. Health Perspect. 119, 1691–1699. https://doi.org/10.1289/ehp.1003369

Jerez Vegueria, S.F., Godoy, J.M., Miekeley, N., 2002. Environmental impact studies of barium and radium discharges by produced waters from the “Bacia de Campos” oil-field offshore platforms, Brazil. J. Environ. Radioact. 62, 29–38. https://doi.org/10.1016/S0265-931X(01)00148-5

Jernelöv, A., 2010. The Threats from Oil Spills: Now, Then, and in the Future. Ambio 39, 353–366. https://doi.org/10.1007/s13280-010-0085-5

Johnson, M.R., Coderre, A.R., 2011. An analysis of flaring and venting activity in the Alberta upstream oil and gas industry. J. Air Waste Manag. Assoc. 61, 190–200. https://doi.org/10.3155/1047-3289.61.2.190

Johnson, M.R., Jefferson, A.M., Corbin, D.J., Conrad, B.M., 2015. Field Measurements of Black Carbon Emissions from Flaring in Coca, Ecuador — Interim Project Report I: Summary of Data Collected during Field Measurements.

Juteau, G., Becerra, S., Maurice, L., 2014. Environment, oil and political vulnerability in the Ecuadorian Amazon: Towards new forms of energy governance? América Lat. Hoy 119–137.

Kaasschieter, E.F., 1999. Solving the Buckley – Leverett equation with gravity in a heterogeneous porous medium. Comput. Geosci. 3Shresh, 23–48. https://doi.org/10.1023/A:1011574824970

Kasperson, J.X., Kasperson, R.E., 2012. Climate Change, Vulnerability and Social Justice. Routledge, pp. 316–336. https://doi.org/10.4324/9781849772549-27

Kasperson, R.E., Matson, P.A., Mccarthy, J.J., Corell, R.W., Christensen, L., Eckley, N., Kasperson, J.X., Luers, A., Martello, M.L., Polsky, C., Pulsipher, A., Schiller, A., 2003. A framework for vulnerability analysis in sustainability science. Elements 100.

Kazakis, N., Voudouris, K.S., 2015. Groundwater vulnerability and pollution risk assessment of porous aquifers to nitrate: Modifying the DRASTIC method using quantitative parameters. J. Hydrol. 525, 13–25. https://doi.org/10.1016/j.jhydrol.2015.03.035

Kessler, M., Abrahamczyk, S., Bos, M., Buchori, D., Putra, D., Gradstein, S.R., Höhn, P., Kluge, J., Orend, F., Pitopang, R., Saleh, S., Schulze, C.H., Sporn, S.G., Kessler, M., Abrahamczyk, S., Bos, M., Buchori, D., Putra, D.D., Tjitrosoedirdjo, S.S., Tscharntke, T., 2016. Alpha and Beta Diversity of Plants and Animals along a Tropical Land-Use Gradient. Ecol. Appl. 19, 2142–2156.

Kimerling, 1990. Disregarding environmental law: petroleum development in protected natural areas and indigenous homelands in the Ecuadorian Amazon. Hast. Int’l Comp. L. Rev. 14, 849–856.

Kimerling, J., 2015. Kimerling, Judith, Habitat as Human Rights: Indigenous Huaorani in the Amazon Rainforest, Oil, and Ome Yasuni., in: Vermont Law Review. pp. 445–524.

Kontovas, C.A., Psaraftis, H.N., Ventikos, N.P., 2010. An empirical analysis of IOPCF oil spill cost data. Mar. Pollut. Bull. 60, 1455–1466. https://doi.org/10.1016/j.marpolbul.2010.05.010

Kuang-Yao Pan, W., Erlien, C., Bilsborrow, R.E., 2010. Morbidity and mortality disparities among colonist and indigenous populations in the Ecuadorian Amazon. Soc. Sci. Med. 70, 401–411. https://doi.org/10.1016/j.socscimed.2009.09.021

Küster, E., Dorusch, F., Altenburger, R., 2005. Effects of hydrogen sulfide to Vibrio fischeri, Scenedesmus vacuolatus, and Daphnia magna. Environ. Toxicol. Chem. 24, 2621–9.

Lacerda, A., Gusmão, G.A., Hamada, N., 2014. Tests of chronic and acute toxicity of crude oil on larvae of Chironomus kiiensis Tokunaga (Diptera: Chironomidae). Braz. J. Biol 74, 70. https://doi.org/10.1590/1519-6984.24012

Lahr, J., Kooistra, L., 2010. Environmental risk mapping of pollutants: State of the art and communication aspects. Sci. Total Environ. 408, 3899–3907. https://doi.org/10.1016/j.scitotenv.2009.10.045

Lahr, J., Münier, B., De Lange, H.J., Faber, J.F., Sørensen, P.B., 2010. Wildlife vulnerability and risk maps for combined pollutants. Sci. Total Environ. 408, 3891–3898. https://doi.org/10.1016/j.scitotenv.2009.11.018

Lal, R., 1997. Degradation and resilience of soils. Philos. Trans. R. Soc. B Biol. Sci. 352, 997–1010. https://doi.org/10.1098/rstb.1997.0078

Laraque, A., Ronchail, J., Cochonneau, G., Pombosa, R., Guyot, J.L., 2007. Heterogeneous Distribution of Rainfall and Discharge Regimes in the Ecuadorian Amazon Basin. J. Hydrometeorol. 8, 1364–1381. https://doi.org/10.1175/2007JHM784.1

Larrea, C., 2006. Hacia una Historia Ecológica del Ecuador: Propuestas para el debate (No. 15), Universidad Andina Simón Bolivar, Biblioteca General de Cultura. Quito.

Larrea, C., Warnars, L., 2009. Ecuador's Yasuni-ITT Initiative: Avoiding emissions by keeping petroleum underground. Energy Sustain. Dev. 13, 219–223. https://doi.org/10.1016/j.esd.2009.08.003

Larsen, K., Delgado, M., Marsters, P., 2015. Untapped Potential: Reducing Global Methane Emissions from Oil and Natural Gas Systems, The Rhodium Group. New York, NY. https://doi.org/10.1089/big.2014.1523

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S., Watt, A.D., 1998. Biodiversity inventories, indicators taxa and effects of habitat modification in tropical forest. Nature 39, 72–76. https://doi.org/10.1603/0046-225X-32.3.542

Legendre, P., Legendre, L., 1998. Numerical Ecology, 2nd Editio. ed, Numerical Ecology. Elsevier B.V., Netherlands.

Lessmann, J., Fajardo, J., Muñoz, J., Bonaccorso, E., 2016. Large expansion of oil industry in the Ecuadorian Amazon: biodiversity vulnerability and conservation alternatives. Ecol. Evol. 6, 4997–5012. https://doi.org/10.1002/ece3.2099

Liber, K., Doig, L.E., White-Sobey, S.L., 2011. Toxicity of uranium, molybdenum, nickel, and arsenic to Hyalella azteca and Chironomus dilutus in water-only and spiked-sediment toxicity tests. Ecotoxicol. Environ. Saf. 74, 1171–1179. https://doi.org/10.1016/j.ecoenv.2011.02.014

Limpert, E., Stahel, W.A., Abbt, M., 2001. Log-normal Distributions across the Sciences: Keys and Clues. Bioscience 51, 341–352. https://doi.org/10.1641/0006-3568(2001)051[0341:lndats]2.0.co;2

Lopes, A., da Rosa-Osman, S.M., Piedade, M.T.F., 2009. Effects of crude oil on survival, morphology, and anatomy of two aquatic macrophytes from the Amazon floodplains. Hydrobiologia 636, 295–305. https://doi.org/10.1007/s10750-009-9959-6

Luers, A.L., Lobell, D.B., Sklar, L.S., Addams, C.L., Matson, P.A., 2003. A method for quantifying vulnerability, applied to the agricultural system of the Yaqui Valley, Mexico. Glob. Environ. Chang. 13, 255–267. https://doi.org/10.1016/S0959-3780(03)00054-2

Maddela, N.R., Burgos, R., Kadiyala, V., Carrion, A.R., Bangeppagari, M., 2016. Removal of petroleum hydrocarbons from crude oil in solid and slurry phase by mixed soil microorganisms isolated from Ecuadorian oil fields. Int. Biodeterior. Biodegradation 108, 85–90. https://doi.org/10.1016/J.IBIOD.2015.12.015

MAE-PRAS, 2016. Pasivos ambientales y reparación ambiental integral: Experiencias de gestion en el Ecuador, 2nd ed. Ministerio del Ambiente, Quito, Ecuador.

MAE, MAGAP, 2015. Mapa de Cobertura y Uso de la Tierra del Ecuador Continental. Quito, Ecuador.

Maes, J., Liquete, C., Teller, A., Erhard, M., Paracchini, M.L., Barredo, J.I., Grizzetti, B., Cardoso, A., Somma, F., Petersen, J.E., Meiner, A., Gelabert, E.R., Zal, N., Kristensen, P., Bastrup-Birk, A., Biala, K., Piroddi, C., Egoh, B., Degeorges, P., Fiorina, C., Santos-Martín, F., Naruševičius, V., Verboven, J., Pereira, H.M., Bengtsson, J., Gocheva, K., Marta-Pedroso, C., Snäll, T., Estreguil, C., San-Miguel-Ayanz, J., Pérez-Soba, M., Grêt-Regamey, A., Lillebø, A.I., Malak, D.A., Condé, S., Moen, J., Czúcz, B., Drakou, E.G., Zulian, G., Lavalle, C., 2016. An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. Ecosyst. Serv. 17, 14–23. https://doi.org/10.1016/j.ecoser.2015.10.023

Maestripieri, N., Saqalli, M., 2016. Assessing health risk using regional mappings based on local perceptions: A comparative study of three different hazards. Hum. Ecol. Risk Assess. 22, 721–735. https://doi.org/10.1080/10807039.2015.1105099

MAGAP-SIGTIERRAS, Tracasa-nipsa, C., 2015. Manual of metodologies for the geopedological study and mapping at scale of 1:25 000. Quito, Ecuador.

Mainville, N., Webb, J., Lucotte, M., Davidson, R., 2006. Decrease of soil fertility and release of mercury following deforestation in the Andean Amazon , Napo River Valley , Ecuador 368, 88–98. https://doi.org/10.1016/j.scitotenv.2005.09.064

Malczewski, J., 2006. GIS-based multicriteria decision analysis: A survey of the literature. Int. J. Geogr. Inf. Sci. 20, 703–726. https://doi.org/10.1080/13658810600661508

Marignani, M., Bruschi, D., Astiaso Garcia, D., Frondoni, R., Carli, E., Pinna, M.S., Cumo, F., Gugliermetti, F., Saatkamp, A., Doxa, A., Queller, E.M., Chaieb, M., Bou Dagher-Kharrat, M., El Zein, R., El Jeitani, S., Khater, C., Mansour, S., Al-Shami, A., Harik, G., Alameddine, I., el-Fadel, M., Blasi, C., 2017. Identification and prioritization of areas with high environmental risk in Mediterranean coastal areas: A flexible approach. Sci. Total Environ. 590–591, 566–578. https://doi.org/10.1016/j.scitotenv.2017.02.221

Martínez, M.L., Pérez-Maqueo, O., Vázquez, G., Castillo-Campos, G., García-Franco, J., Mehltreter, K., Equihua, M., Landgrave, R., 2009. Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. For. Ecol. Manage. 258, 1856–1863. https://doi.org/10.1016/j.foreco.2009.02.023

McEwen, J.D.N., Johnson, M.R., 2012. Black carbon particulate matter emission factors for buoyancy-driven associated gas flares. J. Air Waste Manage. Assoc. 62, 307–321. https://doi.org/10.1080/10473289.2011.650040

McKain, K., Down, A., Raciti, S.M., Budney, J., Hutyra, L.R., Floerchinger, C., Herndon, S.C., Nehrkorn, T., Zahniser, M.S., Jackson, R.B., Phillips, N., Wofsy, S.C., 2015. Methane emissions from natural gas infrastructure and use in the urban region of Boston, Massachusetts. Proc. Natl. Acad. Sci. 112, 1941–1946. https://doi.org/10.1073/pnas.1416261112

Mearns, K., Flin, R., 1995. Risk perception and attitudes to safety by personnel in the offshore oil and gas industry: a review. J. Loss Prev. Process Ind. 8, 299–305. https://doi.org/10.1016/0950-4230(95)00032-V

Medellín, R.A., Equihua, M., 1998. Mammal species richness and habitat use in rainforest and abandoned agricultural fields in Chiapas, Mexico. J. Appl. Ecol. 35, 13–23. https://doi.org/10.1046/j.1365-2664.1998.00272.x

Mena, Barbieri, A.F., Walsh, S.J., Erlien, C.M., Holt, F.L., Bilsborrow, R.E., 2006. Pressure on the Cuyabeno Wildlife Reserve: Development and Land Use/Cover Change in the Northern Ecuadorian Amazon. World Dev. 34, 1831–1849. https://doi.org/10.1016/j.worlddev.2006.02.009

Mena, C., 2008. Trajectories of land-use and land-cover in the northern Ecuadorian Amazon: Temporal composition, spatial configuration, and probability of change. Photogramm. Eng. Remote Sens. 74, 737–751. https://doi.org/10.20472/BM.2015.3.2.005

Mena, C., Bilsborrow, R.E., McClain, M.E., 2006. Socioeconomic drivers of deforestation in the Northern Ecuadorian Amazon. Environ. Manage. 37, 802–815. https://doi.org/10.1007/s00267-003-0230-z

Meng, Q., 2015. Spatial analysis of environment and population at risk of natural gas fracking in the state of Pennsylvania, USA. Sci. Total Environ. 515–516, 198–206. https://doi.org/10.1016/j.scitotenv.2015.02.030

Merchán-Rivera, P., Chiogna, Gabriele, 2017. Assessment of contamination by petroleum hydrocarbons from oil exploration and production activities in Aguarico , Ecuador . Munich, Germany.

Merchant, J.W., 1994. GIS-Base Groundwater Pollution Hazard Assessment: Critical Review of the DRASTIC Model. Photogramm. Eng. Remote Sens. 60, 1117–1127.

Messina, J.P., Walsh, S.J., 2005. Dynamic spatial simulation modeling of the population-environment matrix in the Ecuadorian Amazon. Environ. Plan. B Plan. Des. 32, 835–856. https://doi.org/10.1068/b31186

Messina, J.P., Walsh, S.J., Mena, C.F., Delamater, P.L., 2006. Land tenure and deforestation patterns in the Ecuadorian Amazon: Conflicts in land conservation in frontier settings. Appl.

Geogr. 26, 113–128. https://doi.org/10.1016/j.apgeog.2005.11.003

Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L., Leemans, R., Schröter, D., 2006. The vulnerability of ecosystem services to land use change. Agric. Ecosyst. Environ. 114, 69–85. https://doi.org/10.1016/J.AGEE.2005.11.025

Metzger, P., Ercole, R.D., Metzger, P., Ercole, R.D., 2008. Enjeux territoriaux et vulnerabilité : une approche operationnelle. Colloq. Interdiscip. ” Vulnerabilites sociétales, risques Environ. Compr. les evaluer”, May 2008, Toulouse, Fr.

Miguez-Macho, G., Fan, Y., 2012. The role of groundwater in the Amazon water cycle: 1. Influence on seasonal streamflow, flooding and wetlands. J. Geophys. Res. Atmos. 117, 1–30. https://doi.org/10.1029/2012JD017539

Miller, F., Osbahr, H., Boyd, E., Thomalla, F., Bharwani, S., Ziervogel, G., Walker, B., Birkmann, J., Van der Leeuw, S., Rockström, J., Hinkel, J., Downing, T., Folke, C., Nelson, D., 2010. Resilience and vulnerability: Complementary or conflicting concepts? Ecol. Soc. 15. https://doi.org/10.5751/ES-03378-150311

Mimi, Z.A., Assi, A., 2009. Intrinsic vulnerability, hazard and risk mapping for karst aquifers: A case study. J. Hydrol. 364, 298–310. https://doi.org/10.1016/j.jhydrol.2008.11.008

Mimi, Z.A., Mahmoud, N., Madi, M.A., 2012. Modified DRASTIC assessment for intrinsic vulnerability mapping of karst aquifers: A case study. Environ. Earth Sci. https://doi.org/10.1007/s12665-011-1252-0

MONOIL, 2017. Monitoring environmental, health, society and petroleum in Ecuador [WWW Document]. URL http://www.monoil.ird.fr/ (accessed 9.24.18).

Moolgavkar, S.H., Chang, E.T., Watson, H., Lau, E.C., 2014. Cancer mortality and quantitative oil production in the Amazon region of Ecuador, 1990-2010. Cancer Causes Control 25, 59–72. https://doi.org/10.1007/s10552-013-0308-8

Moser, G., Hertel, D., Leuschner, C., 2007. Altitudinal Change in LAI and Stand Leaf Biomass in Tropical Montane Forests: a Transect Study in Ecuador and a Pan-Tropical Meta-Analysis. Ecosystems 10, 924–935. https://doi.org/10.1007/s10021-007-9063-6

MSS, 2015. National Energetic Balance. Quito, Ecuador.

MSS, 2014. National Energetic Balance. Quito, Ecuador. https://doi.org/10.1017/CBO9781107415324.004

MSS, 2013. National Energetic Balance. Quito, Ecuador.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–8. https://doi.org/10.1038/35002501

Napolitano, P., Fabbri, A.G., 1996. Single-parameter sensitivity analysis for aquifer vulnerability assessment using DRASTIC and SINTACS. IAHS Publ. Proc. Reports-Intern Assoc Hydrol. Sci. 235, 559–566.

National Board of Hydrocarbons, 2016. Exploration: production subprocess oil wells sampling by oil operator.

Natural Research Council, 1993. Ground Water Vulnerability Assessment. National Academies

Press. https://doi.org/10.17226/2050

NBH, 2018. Biblioteca – Secretaría de Hidrocarburos [WWW Document]. Quito, Ecuador. URL http://www.secretariahidrocarburos.gob.ec/biblioteca/ (accessed 9.21.18).

Nebeker, A.V., Schuytema, G.S., Ott, S.L., 1995. Effects of cadmium on growth and bioaccumulation in the northwestern salamander Ambystoma gracile. Arch. Environ. Contam. Toxicol. 29, 492–499. https://doi.org/10.1007/BF00208379

Neergheen-Bhujun, V., Awan, A.T., Baran, Y., Bunnefeld, N., Chan, K., Dela Cruz, T.E., Egamberdieva, D., Elsässer, S., Johnson, M.-V. V, Komai, S., Konevega, A.L., Malone, J.H., Mason, P., Nguon, R., Piper, R., Shrestha, U.B., Pešić, M., Kagansky, A., 2017. Biodiversity, drug discovery, and the future of global health: Introducing the biodiversity to biomedicine consortium, a call to action. J. Glob. Health 7, 020304. https://doi.org/10.7189/jogh.07.020304

Ngo, V. V, Michel, J., Gujisaite, V., Latifi, A., Simonnot, M.-O., 2014. Parameters describing nonequilibrium transport of polycyclic aromatic hydrocarbons through contaminated soil columns: estimability analysis, correlation, and optimization. J. Contam. Hydrol. 158, 93–109. https://doi.org/10.1016/j.jconhyd.2014.01.005

Nordberg, G., 2007. Handbook on the toxicology of metals.

Ochoa-Cueva, P., Fries, A., Montesinos, P., Rodríguez-Díaz, J.A., Boll, J., 2015. Spatial Estimation of Soil Erosion Risk by Land-cover Change in the Andes OF Southern Ecuador. L. Degrad. Dev. 26, 565–573. https://doi.org/10.1002/ldr.2219

OECD, 2016. Crude Oil Production [WWW Document]. Energy Stat. OECD Ctries. URL https://data.oecd.org/energy/crude-oil-production.htm (accessed 9.19.18).

Oliver, T.H., Heard, M.S., Isaac, N.J.B., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C.D.L., Petchey, O.L., Proença, V., Raffaelli, D., Suttle, K.B., Mace, G.M., Martín-López, B., Woodcock, B.A., Bullock, J.M., 2015. Biodiversity and Resilience of Ecosystem Functions. Trends Ecol. Evol. 30, 673–684. https://doi.org/10.1016/j.tree.2015.08.009

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'amico, J. a., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P., Kassem, K.R., 2001. Terrestrial Ecoregions of the World: A New Map of Life on Earth. Bioscience 51, 933. https://doi.org/10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2

Open Knowledge International, 2015. Global Open Data Index [WWW Document]. Cambridge, United Kingdom. URL http://2015.index.okfn.org/place/2014/ (accessed 4.24.18).

Otieno-odawa, C.F., Kaseje, D.O., 2014. Validity and reliability of data collected by community health workers in rural and peri- urban contexts in Kenya. BMC Health Serv. Res. 14, S5. https://doi.org/10.1186/1472-6963-14-S1-S5

Pacheco, F.A.L., Pires, L.M.G.R., Santos, R.M.B., Sanches Fernandes, L.F., 2015. Factor weighting in DRASTIC modeling. Sci. Total Environ. 505, 474–486. https://doi.org/10.1016/j.scitotenv.2014.09.092

Palermo, F., Parra, R.., 2014. Procesamiento de parámetros físicos e inventario de emisiones atmosféricas antropogénicas en la cuenca hidrográfica del río Napo. Universidad San Francisco de Quito.

Pan, W.K.., Walsh, S.J., Bilsborrow, R.E., Frizzelle, B.G., Erlien, C.M., Baquero, F., 2004. Farm-level models of spatial patterns of land use and land cover dynamics in the Ecuadorian Amazon. Agric. Ecosyst. Environ. 101, 117–134. https://doi.org/10.1016/j.agee.2003.09.022

Pappalardo, S.E., De Marchi, M., Ferrarese, F., 2013. Uncontacted Waorani in the Yasuní Biosphere Reserve: Geographical Validation of the Zona Intangible Tagaeri Taromenane (ZITT). PLoS One 8, e66293. https://doi.org/10.1371/journal.pone.0066293

Patrick, R., Bott, T., Larson, R., 1975. The role of trace elements in management of nuisance growth.

PBRP, 1992. Conserving Biodiversity: A Research Agenda for Development Agencies, Development. Washington, DC.

Peláez-Samaniego, M.R., Garcia-Perez, M., Cortez, L.A.B., Oscullo, J., Olmedo, G., 2007. Energy sector in Ecuador: Current status. Energy Policy 35, 4177–4189. https://doi.org/10.1016/j.enpol.2007.02.025

Pinheiro, A., 2008. Assessing Risk: A Simplified Methodology for Prejob Planning in Oil & Gas Production. Prof. Saf. 53, 34–41.

Pressey, R.L., Cabeza, M., Watts, M.E., Cowling, R.M., Wilson, K.A., 2007. Conservation planning in a changing world. Trends Ecol. Evol. 22, 583–592. https://doi.org/10.1016/J.TREE.2007.10.001

Preston, B.L., Yuen, E.J., Westaway, R.M., 2011. Putting vulnerability to climate change on the map: a review of approaches, benefits, and risks. Sustain. Sci. 6, 177–202. https://doi.org/10.1007/s11625-011-0129-1

Province GAD Orellana, 2011. Land Use and Management Plan for Orellana. El Coca, Orellana.

Province GAD Sucumbios, 2013. Land Use and Management Plan for Sucumbios. Sucumbios, Ecuador.

PSB-MAE, 2013. Programa Socio Bosque [WWW Document]. Programas y Proy. URL http://sociobosque.ambiente.gob.ec/node/173 (accessed 9.22.18).

PUCE, 2018. Museums of Biological Sciences [WWW Document]. Pontif. Univ. Catol. del Ecuador. URL https://www.puce.edu.ec/portal/content/Museo de Zoología QCAZ Vertebrados/482?link=oln30.redirect (accessed 10.18.18).

Rahman, A., 2008. A GIS based DRASTIC model for assessing groundwater vulnerability in shallow aquifer in Aligarh, India. Appl. Geogr. 28, 32–53. https://doi.org/10.1016/j.apgeog.2007.07.008

Ramanathan, V., Carmichael, G., 2008. Global and regional climate changes due to black carbon. Nat. Geosci. 1, 221–227. https://doi.org/10.1038/ngeo156

Rausand, M., 2011. Risk Assessment: Theory, Methods, and Applications. https://doi.org/10.1002/9781118281116

Reátegui-Zirena, E.G., Stewart, P.M., Whatley, A., Chu-Koo, F., 2012. Acute Toxicity and Mutagenicity of Peruvian Crude Oil and Oil-Contaminated Samples from the Peruvian

Amazon Final Report.

Rehr, Amanda P; Small, Mitchel J; Mathews Scott H; Hendrickson, C.T., 2010. Economic Sources and Spatial Distribution of Airborne Chromium Risks in the U . S . Environ. Sci. Technol. 44, 2131–2137.

Reinke, D., 2011. The species-specific effects of sublethal concentrations of cadmium on freshwater phytoplankton communities in a Canadian Shield lake. Artic. Can. J. Bot. https://doi.org/10.1139/b85-279

Resilience Alliance and Santa Fe Institute, 2018. Resilience Alliance [WWW Document]. URL https://www.resalliance.org/thresholds-db (accessed 10.17.18).

Ribeiro, L., Pindo, J.C., Dominguez-Granda, L., 2017. Assessment of groundwater vulnerability in the Daule aquifer, Ecuador, using the susceptibility index method. Sci. Total Environ. 574, 1674–1683. https://doi.org/10.1016/j.scitotenv.2016.09.004

Robalino-López, A., Mena-Nieto, A., García-Ramos, J.E., 2014. System dynamics modeling for renewable energy and CO2 emissions: A case study of Ecuador. Energy Sustain. Dev. 20, 11–20. https://doi.org/10.1016/J.ESD.2014.02.001

Rosário, F.F. do, Custodio, E., Silva, G.C. da, 2016. Hydrogeology of the Western Amazon Aquifer System (WAAS). J. South Am. Earth Sci. 72, 375–386. https://doi.org/10.1016/j.jsames.2016.10.004

Roy, D.C., Blaschke, T., 2015. Spatial vulnerability assessment of floods in the coastal regions of Bangladesh. Geomatics, Nat. Hazards Risk 6, 21–44. https://doi.org/10.1080/19475705.2013.816785

Rwengabo, S., 2018. Efficiency, Sustainability and Exit Strategy in the Oil and Gas Sector: Lessons from Ecuador for Uganda. Advocates Coalition for Development and Environment (ACODE).

Sala, O.E., Iii, F.S.C., Armesto, J.J., Berlow, E., Dirzo, R., Huber-sanwald, E., Huenneke, L.F., Robert, B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global Biodiversity Scenarios for the Year 2100 Global Biodiversity Scenarios for the Year 2100. Science (80-.). 287, 1770–1774. https://doi.org/10.1126/science.287.5459.1770

San Sebastián, M., Armstrong, B., Córdoba, J.A., Stephens, C., 2001. Exposures and cancer incidence near oil fields in the Amazon basin of Ecuador. Occup. Environ. Med. 58, 517–522. https://doi.org/10.1136/oem.58.8.517

San Sebastián, M., Hurtig, A.K., 2005. Oil development and health in the Amazon basin of Ecuador: the popular epidemiology process. Soc. Sci. Med. 60, 799–807. https://doi.org/10.1016/j.socscimed.2004.06.016

Saqalli, M., Béguet, E., Maestripieri, N., Morin, L., Cabrero, F., De Garine, E., 2018. “Somos Amazonia”, a new inter-indigenous identity in Ecuadorian Amazonia: toward a Jus aplidia, a right based on ecological origin?

Saqalli, M., Jourdren, M., Maestripieri, N., Guillerme, S., Soulileuth, B., Latsachach, K., Sounyafong, P., Tammahuxsa, L., Sengtaheuanghoung, O., Saqalli, M., Jourdren, M., Maestripieri, N., Guillerme, S., 2015. Backward waters , modern waters : Perception-Based Regional Mapping territory uses and water-related sanitary stakes in Luang Phabang area (Lao PDR) 184–193. https://doi.org/10.1016/j.apgeog.2015.04.001

Sarah, a., Julius, U., Francis, C., 2015. Determination of Some Petrophysical Properties of Reservoir Rocks in the Niger Delta. J. Sci. Res. Reports 5, 388–401. https://doi.org/10.9734/JSRR/2015/15096

Sax, T., Isakov, V., 2003. A case study for assessing uncertainty in local-scale regulatory air quality modeling applications. Atmos. Environ. 37, 3481–3489. https://doi.org/10.1016/S1352-2310(03)00411-4

Saxton, K.E., Rawls, W.J., 2006. Soil Water Characteristic Estimates by Texture and Organic Matter for Hydrologic Solutions. Soil Sci. Soc. Am. J. 70, 1569. https://doi.org/10.2136/sssaj2005.0117

Schmidt Etkin, D., 2001. Analysis Of Oil Spill Trends In The United States And Worldwide, in: [1] D. Schmidt Etkin, Analysis Of Oil Spill Trends In The United States And Worldwide, (n.d.). Http://Www.Environmental-Research.Com/Publications/Pdf/Spill_statistics/Paper4.Pdf (Accessed August 15, 2017).

Schmidt, J.H., 2008. Development of LCIA characterisation factors for land use impacts on biodiversity. J. Clean. Prod. 16, 1929–1942. https://doi.org/10.1016/J.JCLEPRO.2008.01.004

Schulze, C.H., Waltert, M., Kessler, P.J.A., Pitopang, R., Leuschner, C., Steffan-dewenter, I., Tscharntke, T., 2017. Biodiversity Indicator Groups of Tropical Land-Use Systems : Comparing Plants , Birds , and Insects Leuschner , Ingolf Steffan-Dewenter and Teja Tscharntke Source : Ecological Applications , Vol . 14 , No . 5 (Oct ., 2004), pp . 1321-1333 Published by :

Schweitzer, L., 2008. Accident frequencies in environmental justice assessment and land use studies. J. Hazard. Mater. 156, 44–50. https://doi.org/10.1016/j.jhazmat.2007.11.125

Seabra, V.S., Da Silva, G.C., Cruz, C.B.M., 2009. The use of geoprocessing to assess vulnerability on the east coast aquifers of Rio de Janeiro State, Brazil. Environ. Geol. 57, 665–674. https://doi.org/10.1007/s00254-008-1345-6

Secunda, S., Collin, M.., Melloul, A.., 1998. Groundwater vulnerability assessment using a composite model combining DRASTIC with extensive agricultural land use in Israel’s Sharon region. J. Environ. Manage. 54, 39–57. https://doi.org/10.1006/jema.1998.0221

Selby, D.A., Ihnat, J.M., Messer, J.J., 1985. Effects of subacute cadmium exposure on a hardwater mountain stream microcosm. Water Res. 19, 645–655. https://doi.org/10.1016/0043-1354(85)90071-5

SENAGUA-ESPOL, 2017. Hidrología y plan hidráulico regional de la demarcación hidrográfica de Napo. Quito, Ecuador.

SENPLADES, 2013. Atlas de las desigualdades socioeconómicas del Ecuador, TRAMA. ed. Quito.

Serra-Sogas, N., O'Hara, P.D., Canessa, R., Keller, P., Pelot, R., 2008. Visualization of spatial patterns and temporal trends for aerial surveillance of illegal oil discharges in western Canadian marine waters. Mar. Pollut. Bull. 56, 825–833. https://doi.org/10.1016/j.marpolbul.2008.02.005

Shahriar, A., Sadiq, R., Tesfamariam, S., 2012. Risk analysis for oil & gas pipelines: A sustainability assessment approach using fuzzy based bow-tie analysis. J. Loss Prev. Process Ind. 25, 505–523. https://doi.org/10.1016/j.jlp.2011.12.007

Shindell, D., Kuylenstierna, J.C.I., Vignati, E., van Dingenen, R., Amann, M., Klimont, Z., Anenberg, S.C., Muller, N., Janssens-Maenhout, G., Raes, F., Schwartz, J., Faluvegi, G., Pozzoli, L., Kupiainen, K., Hoglund-Isaksson, L., Emberson, L., Streets, D., Ramanathan, V., Hicks, K., Oanh, N.T.K., Milly, G., Williams, M., Demkine, V., Fowler, D., 2012. Simultaneously Mitigating Near-Term Climate Change and Improving Human Health and Food Security. Science (80-.). 335, 183–189. https://doi.org/10.1126/science.1210026

Shrestha, S., Kafle, R., Pandey, V.P., 2017. Evaluation of index-overlay methods for groundwater vulnerability and risk assessment in Kathmandu Valley, Nepal. Sci. Total Environ. 575, 779–790. https://doi.org/10.1016/j.scitotenv.2016.09.141

Sierra, R., 2000. Dynamics and patterns of deforestation in the western Amazon: The Napo deforestation front, 1986-1996. Appl. Geogr. 20, 1–16. https://doi.org/10.1016/S0143-6228(99)00014-4

Sierra, R., Campos, F., Chamberlin, J., 2002. Assessing biodiversity conservation priorities: Ecosystem risk and representativeness in continental Ecuador. Landsc. Urban Plan. 59, 95–110. https://doi.org/10.1016/S0169-2046(02)00006-3

Silbert, S.F., 2002. From shade - to sun - grow perrennial crops in Sulawesi, Indonesia: Implications for biodiversity conservation and soil fertility. Biodivers. Conserv. 11, 1889–1902. https://doi.org/10.1023/A:102080461

Simpson, I.J., Sulbaek Andersen, M.P., Meinardi, S., Bruhwiler, L., Blake, N.J., Helmig, D., Rowland, F.S., Blake, D.R., 2012. Long-term decline of global atmospheric ethane concentrations and implications for methane. Nature 488, 490–494. https://doi.org/10.1038/nature11342

Sindelarova, K., Granier, C., Bouarar, I., Guenther, A., Tilmes, S., Stavrakou, T., Müller, J.-F., Kuhn, U., Stefani, P., Knorr, W., 2014. Global data set of biogenic VOC emissions calculated by the MEGAN model over the last 30 years. Atmos. Chem. Phys. 14, 9317–9341. https://doi.org/10.5194/acp-14-9317-2014

Singh, K., Chandra, S., 2014. Treatment of Petroleum Hydrocarbon Polluted Environment Through Bioremediation: A Review. Pakistan J. Biol. Sci. 17, 1–8. https://doi.org/10.3923/pjbs.2014.1.8

Smit, B., Wandel, J., 2006. Adaptation, adaptive capacity and vulnerability. Glob. Environ. Chang. 16, 282–292. https://doi.org/10.1016/j.gloenvcha.2006.03.008

Solov, A., 2011. Associated petroleum gas flaring: Environmental issues. Russ. J. Gen. Chem. 81, 2531–2541. https://doi.org/10.1134/S1070363211120218

Soltanieh, M., Zohrabian, A., Gholipour, M.J., Kalnay, E., 2016. A review of global gas flaring and venting and impact on the environment: Case study of Iran. Int. J. Greenh. Gas Control 49, 488–509. https://doi.org/10.1016/j.ijggc.2016.02.010

Sourdat, M., Custode, E., 1982. ESTUDIO MORFO-EDARFOLOGICO -Memoria Técica-Prov. Napo.pdf.

Sovacool, B.K., Scarpaci, J., 2016. Energy justice and the contested petroleum politics of stranded assets: Policy insights from the Yasuní-ITT Initiative in Ecuador. Energy Policy 95, 158–171. https://doi.org/10.1016/J.ENPOL.2016.04.045

Spearman, C., 1907. Demonstration of Formulae for True Measurement of Correlation. Source Am. J. Psychol. 18, 161–169.

Spehar, R.L., Fiandt, J.T., 1986. Acute and chronic effects of water quality criteria-based metal mixtures on three aquatic species. Environ. Toxicol. Chem. 5, 917–931. https://doi.org/10.1002/etc.5620051008

SRA, 2015. Risk Analysis Foundations. Herndon, VA.

Stigter, T.Y., Ribeiro, L., Dill, A.M.M.C., 2006. Evaluation of an intrinsic and a specific vulnerability assessment method in comparison with groundwater salinisation and nitrate contamination levels in two agricultural regions in the south of Portugal. Hydrogeol. J. 14, 79–99. https://doi.org/10.1007/s10040-004-0396-3

Streets, D.G., Gupta, S., Waldhoff, S.T., Qang, M.Q., Bond, T.C., Bo, Y., 2001. Black carbon emissions in China. Atmos. Environ. 35, 4281–4296.

Thirumalaivasan, D., Karmegam, M., Venugopal, K., 2003. AHP-DRASTIC: software for specific aquifer vulnerability assessment using DRASTIC model and GIS. Environ. Model. Softw. 18, 645–656. https://doi.org/10.1016/S1364-8152(03)00051-3

Tirkey, P., Gorai, A.K., Iqbal, J., 2013. AHP-GIS Based DRASTIC Model for Groundwater Vulnerability to Pollution Assessment : A Case Study of Hazaribag District , Jharkhand , India. Int. J. Environ. Prot. 2, 20–31.

Turner, B.L., Kasperson, R.E., Matson, P.A., McCarthy, J.J., Corell, R.W., Christensen, L., Eckley, N., Kasperson, J.X., Luers, A., Martello, M.L., Polsky, C., Pulsipher, A., Schiller, A., 2003. A framework for vulnerability analysis in sustainability science. Proc Natl Acad Sci U S A 100, 8074–8079. https://doi.org/10.1073/pnas.1231335100

UNESCO, 2005. Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis. Washington, DC.

United Nations, 2004. Living with risk: a global review of disaster reduction initiatives, Strategy. United Nations, New York and Geneva. https://doi.org/9211010640

US. Department of Energy, 2017. Natural Gas Spot and Future Prices [WWW Document]. URL http://tonto.eia.gov/dnav/ng/hist/rngc1d.htm%0A (accessed 9.4.17).

Vallejo, M.C., Burbano, R., Falconí, F., Larrea, C., 2015. Leaving oil underground in Ecuador: The Yasuní-ITT initiative from a multi-criteria perspective. Ecol. Econ. 109, 175–185. https://doi.org/10.1016/J.ECOLECON.2014.11.013

Van Straalen, N.M., 1993. Biodiversity of Ecotoxicological Responses in Animals. Netherlands J. Zool. 44, 112–129. https://doi.org/10.1163/156854294X00097

Vanos, J.K., Cakmak, S., Bristow, C., Brion, V., Tremblay, N., Martin, S.L., Sheridan, S.S., 2013. Synoptic weather typing applied to air pollution mortality among the elderly in 10 Canadian cities. Environ. Res. 126, 66–75. https://doi.org/10.1016/J.ENVRES.2013.08.003

Velma, V., Vutukuru, S.S., Tchounwou, P.B., 2009. Ecotoxicology of hexavalent chromium in freshwater fish: a critical review. Rev. Environ. Health 24, 129–45.

Vighi, M., 1981. Lead uptake and release in an experimental trophic chain. Ecotoxicol. Environ. Saf. 5, 177–193. https://doi.org/10.1016/0147-6513(81)90033-6

Vogt, 2015. Vulnerability of tropical forest ecosystems and forest dependent communities to droughts. https://doi.org/10.1016/j.envres.2015.10.022

Vogt, D.J., Vogt, K.A., Gmur, S.J., Scullion, J.J., Suntana, A.S., Daryanto, S., Sigurdardóttir, R., 2016. Vulnerability of tropical forest ecosystems and forest dependent communities to droughts. Environ. Res. 144, 27–38. https://doi.org/10.1016/j.envres.2015.10.022

Vrba, J., Zaporozec, A., 1994. Guidebook on mapping groundwater vulnerability, 16th ed. H.Heise, Hannover.

Wake, H., 2005. Oil refineries: A review of their ecological impacts on the aquatic environment. Estuar. Coast. Shelf Sci. 62, 131–140. https://doi.org/10.1016/j.ecss.2004.08.013

Waldner, C., 2008. The association between exposure to the oil and gas industry and beef calf mortality in Western Canada. Arch. Environ. Occup. Heal. 63, 220–240. https://doi.org/10.3200/AEOH.63.4.220-240

Waldner, C.L., Ribble, C.S., Janzen, E.D., Campbell, J.R., 2001. Associations between oil- and gas-well sites, processing facilities, flaring, and beef cattle reproduction and calf mortality in western Canada. Prev. Vet. Med. 50, 1–17. https://doi.org/10.1016/S0167-5877(01)00214-8

Walker, B., Carpenter, S., Anderies, J., Abel, N., Cumming, G., Janssen, M., Norberg, J., Peterson, G.D., Pritchard, R., 2002. Resilience Management in Social-ecological Systems: a Working Hypothesis for a Participatory Approach. Conserv. Biol. 6.

Walsh, S.J., Messina, J.P., Mena, C.F., Malanson, G.P., Page, P.H., 2008a. Complexity theory, spatial simulation models, and land use dynamics in the Northern Ecuadorian Amazon. Geoforum 39, 867–878. https://doi.org/10.1016/j.geoforum.2007.02.011

Walsh, S.J., Shao, Y., Mena, C.F., McCleary, a L., 2008b. Integration of hyperion satellite data and a household social survey to characterize the causes and consequences of reforestation patterns in the northern Ecuadorian Amazon. Photogramm. Eng. Remote Sensing 74, 725–735.

Wasserstrom, R., 2013. Deforestation, Agrarian Reform and Oil Development in Ecuador, 1964-1994. Nat. Resour. 04, 31–44. https://doi.org/10.4236/nr.2013.41004

Webb, J., Coomes, O.T., Mainville, N., Mergler, D., 2015. Mercury Contamination in an Indicator Fish Species from Andean Amazonian Rivers Affected by Petroleum Extraction. Bull. Environ. Contam. Toxicol. 95, 279–285. https://doi.org/10.1007/s00128-015-1588-3

Weigand, H., Totsche, K.U., Huwe, B., Kögel-Knabner, I., 2001. PAH mobility in contaminated industrial soils: a Markov chain approach to the spatial variability of soil properties and PAH levels. Geoderma 102, 371–389. https://doi.org/10.1016/S0016-7061(01)00043-X

Wernersson, A.-S., 2004. Aquatic ecotoxicity due to oil pollution in the Ecuadorian Amazon. Aquat. Ecosyst. Health Manag. 7, 127–136. https://doi.org/10.1080/14634980490281470

World Bank Group, 2018. Gas Flaring Reduction Partnership [WWW Document]. Gas flaring data 2013-16. URL http://www.worldbank.org/en/programs/gasflaringreduction (accessed 8.24.17).

WorldBank, 2004. Regulation of Associated Gas Flaring and Venting: A Global Overview and Lessons from International Experience.

Zar, J., 1974. Biostatistical Analysis. PsycCRITIQUES 19, 663. https://doi.org/10.1017/CBO9781107415324.004

Zghibi, A., Merzougui, A., Chenini, I., Ergaieg, K., Zouhri, L., Tarhouni, J., 2016. Groundwater vulnerability analysis of Tunisian coastal aquifer: An application of DRASTIC index method in GIS environment. Groundw. Sustain. Dev. 2–3, 169–181. https://doi.org/10.1016/j.gsd.2016.10.001

VIII. Appendix

Section A. Rapid on-line survey.

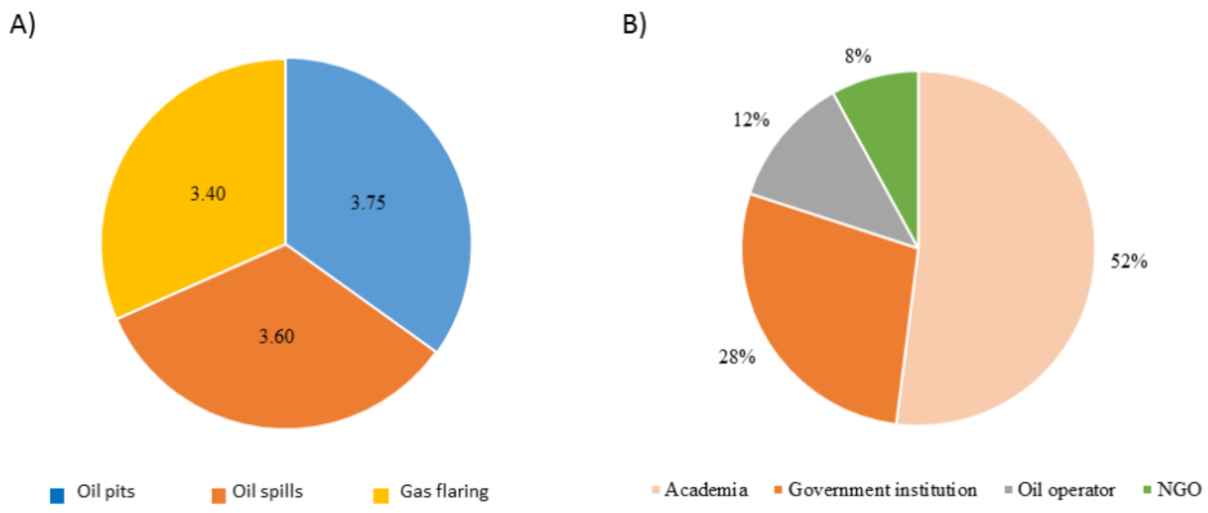
A1. Matrix for question 1 with categories that indicate the effects of contaminants in regard to the negative impacts to the total environment, in other words, to biodiversity, crop yields, water sources and natural resources that maintain ecological functions and provide ecosystem services to local populations. The values of the matrix were given classes (that was visible to the respondents), corresponding to a quantity (not seen by the respondents) defined to weight the responses, in an increasing order: low (1), medium low (2), medium (3), medium high (4), and high (5). Subsequently, weights in each class are multiplied by the number of responses for the given class, and their sum is divided by the total number of responses. This provides a weighted average that could gradually vary from a value of 1 (all respondents check the "low" box) or a maximum value of 5 (all respondents check the "high" box).

The Equation is as follows:

$$\text{Weighted average} = \frac{\mathrm{x1w1+x2w2+x3w3+x4w4+x5w5}}{\text{total number of responses}}$$

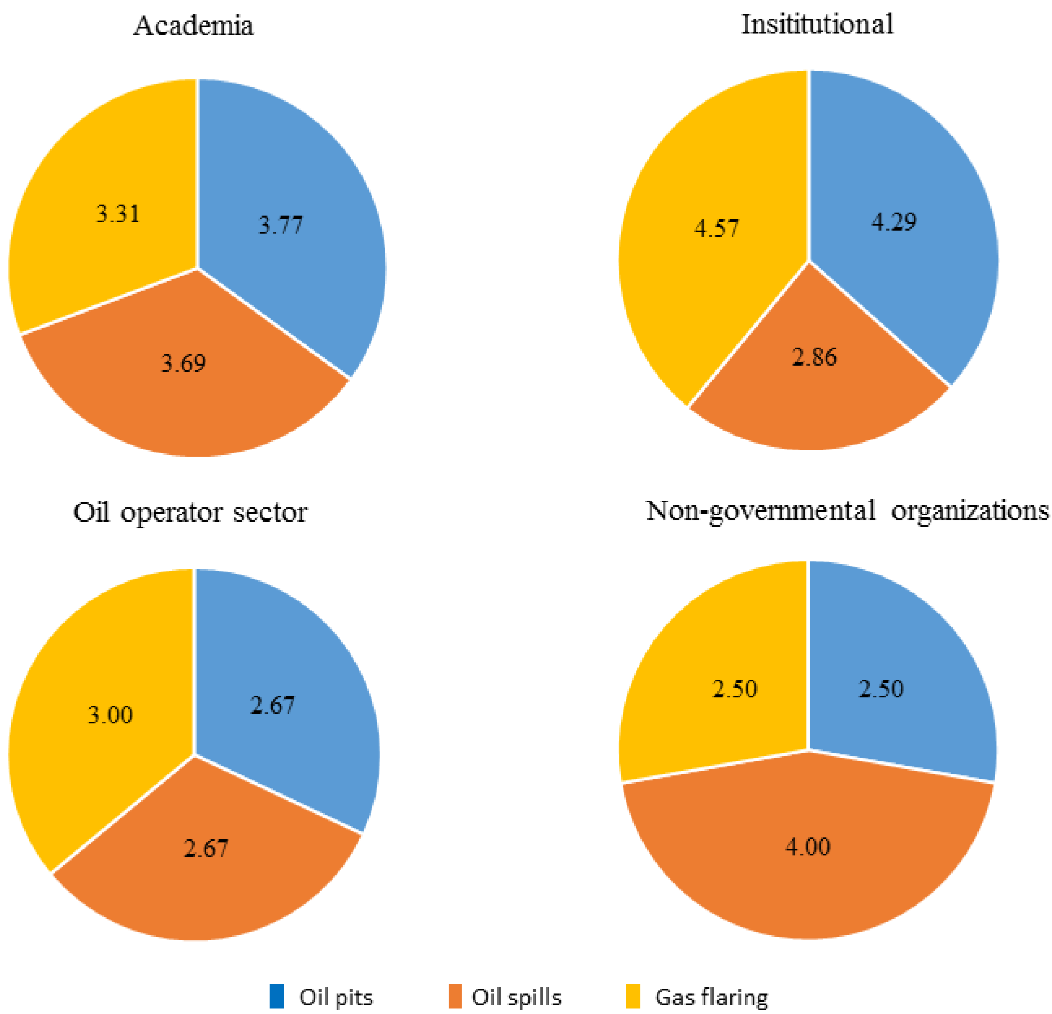
With, w = weights of the given category

x = number of responses in a given category



A2. A) Average weights given to the different contamination sources according to survey responses including all stakeholders; B) percentage of responses by individual stakeholder (N=25) according to their activity sector.

Responses from all stakeholders indicate slightly higher pollution potential from oil pits (3.75), oil spills (3.6) and gas flaring (3.4) without differentiating the number of respondents from each stakeholder. Obviously, the differences were not significant, and the weight for a potential source was probably affected by the number of responses from the stakeholder group (Fig 5.B). Therefore the same procedure was repeated including only respondents of same stakeholder.



A3. Average weights given to the various contaminant sources, disaggregated by stakeholder.

Figure A2 shows there is no consensus amongst respondents regarding the most polluting source. The institutional sector and oil operators comprising mostly public governmental entities identifies gas flaring as the greatest polluting source, non-governmental organizations think oil spills, and academia sector perceived oil pits as the most significant source of pollution. Academia sector being the sector with more respondents it accounted mostly to the final result where oil pits are perceived as having slightly more potential impact on the total hazard.

Section B. Vulnerability of natural heritage to anthropic hazards.

Table B1. Detailed definitions of the heritage sites as defined in the original regulatory institutions.

Heritage site type	Definition
SNAP, Intangible areas	*National System of Protected Areas* (SNAP) is a network of protected areas that contains fifty protected areas at the national level, including three in the NEA: Limoncocha Biological Reserve, Cuyabeno Wildlife Reserve and Yasuní National Park. Limoncocha is also a reserve listed under the International Convention on Wetlands (RAMSAR). While national parks, wilderness areas and biological reserves differ in their definition and objectives yet they share certain criteria: none to limited human presence, maintaining of ecological and hydrological functioning process, integrity and relative uniqueness and threat to their biodiversity (IUCN, 2016). *Intangible Zone (ZITT)* created in 1999, yet only demarcated geographically in 2007, intends to protect two voluntarily isolated indigenous groups, Tagaeri and Taromenane including ancient territories were extractive activities, including forestry are strictly prohibited (Finer et al., 2009, 2008; Vallejo et al., 2014), exceptionally exploitable if natural resources considered of public interest (Finer et al., 2008; Vallejo et al., 2014).
Protected forests	Private, communal or public owned forests, with recreational, touristic, and scientific research purposes that are recognize by local decentralized governments (GAD) but not the SNAP. Several areas are included such as: Sacha Lodge, Unidad 6 Napo, the Scientific Stations San Carlos and Payamino, Cuembi, Cerro Sumaco y Cuenca Alta del Rio Suno and Panacocha.
Biosphere reserves, Forest patrimony	*Patrimony forest* are restricted land category (which were once part of Cuyabeno Wildlife Reserve, succumbed to immigration pressures), where colonists can have communal titles to the lands, limited extractive activities such as farming or timber extraction, it cannot be private owned or open sold to the market (Mena et al., 2006). They serve as buffer areas to larger reserves. *Biosphere reserves* are part of the UNESCO initiative of 1971 that attempts to create sustainable areas where human activities are allowed under sustainable practices. Outpacing traditional confined conservation zones, through appropriate zoning schemes combining core protected areas with zones where sustainable development is fostered by local dwellers and enterprises with often highly innovative and participative governance systems. While education, research, monitoring and capacity enhancement are seen as components of the logistic or knowledge-

	generation function of biosphere reserves, they are also integral to the conservation and development functions (UNESCO, 2017).
Socio-Bosque Programme	*The Socio Bosque Program* (PSB) was created in 2008 to provide economic incentives to private and communal owned forested lands for at least 20 years (http://sociobosque.ambiente.gob.ec/node/174).
Zones without protected status	Mainly all other land use areas that are dedicated to agriculture, urbanization, extractive activities, and any other socioeconomic activity that is not regulated by the MAE, and does not aim to scientific research, conservation and protection of biodiversity or ecosystem functional process sensu-stricto, whereas might need regulation through environmental impact assessments (EIA).

Section C. Risk of groundwater contamination by unlined oil pits.

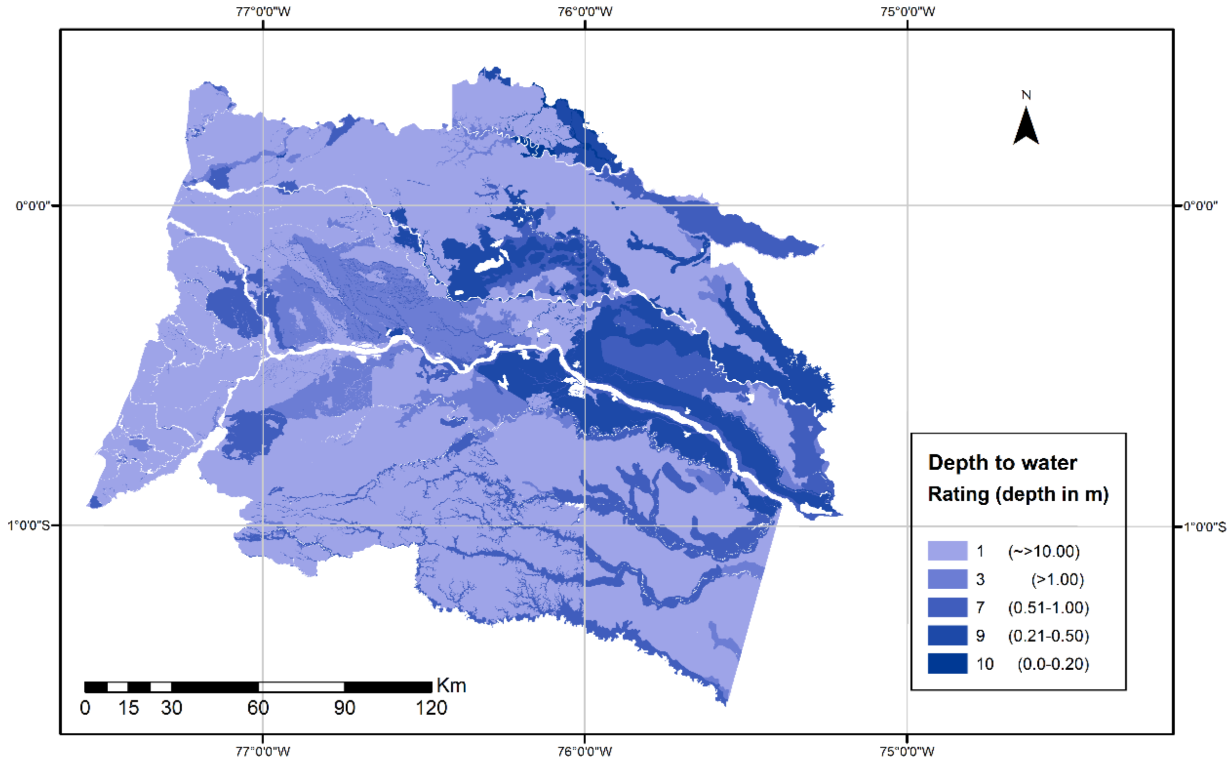


Fig C1. Depth to water showing values measured and given ratings.

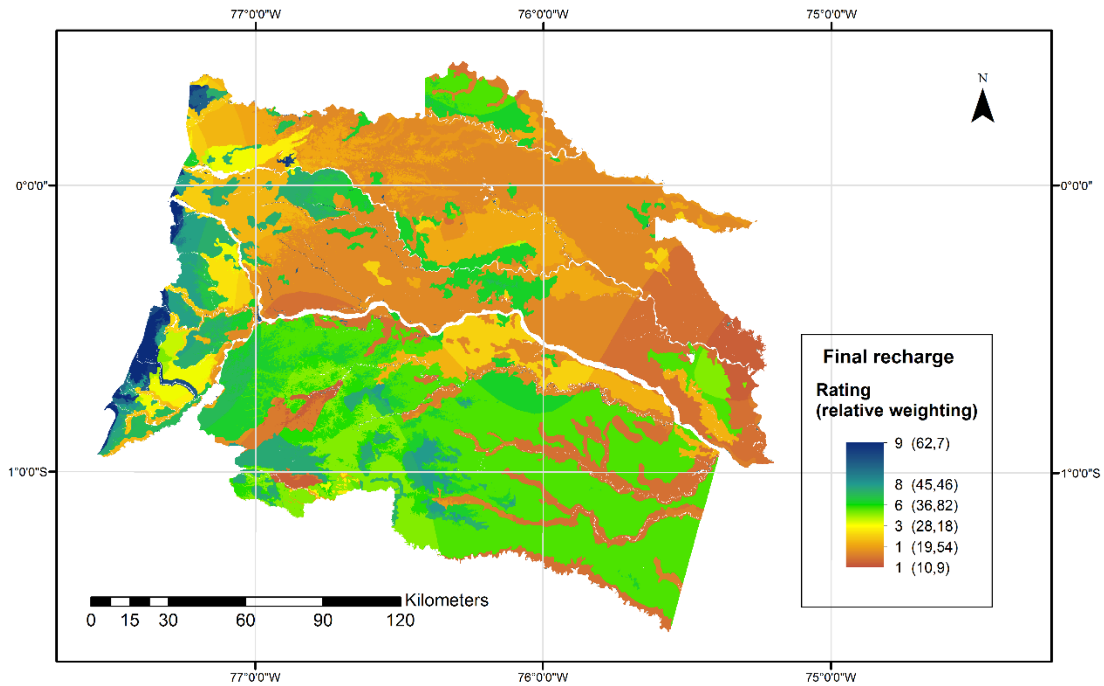


Fig C2. Final recharge approximated by sub-model with corresponding relative weighting and given ratings.

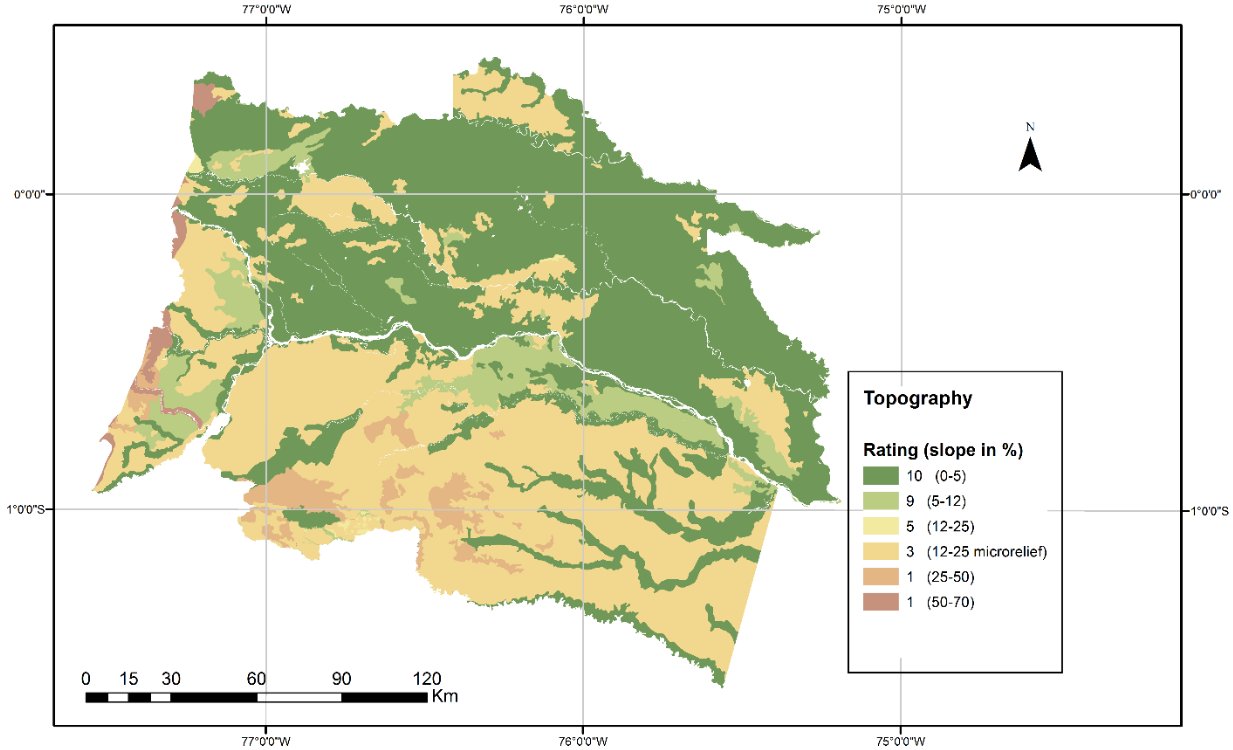


Fig C3. Topography with respective percentage slope and rating scores.

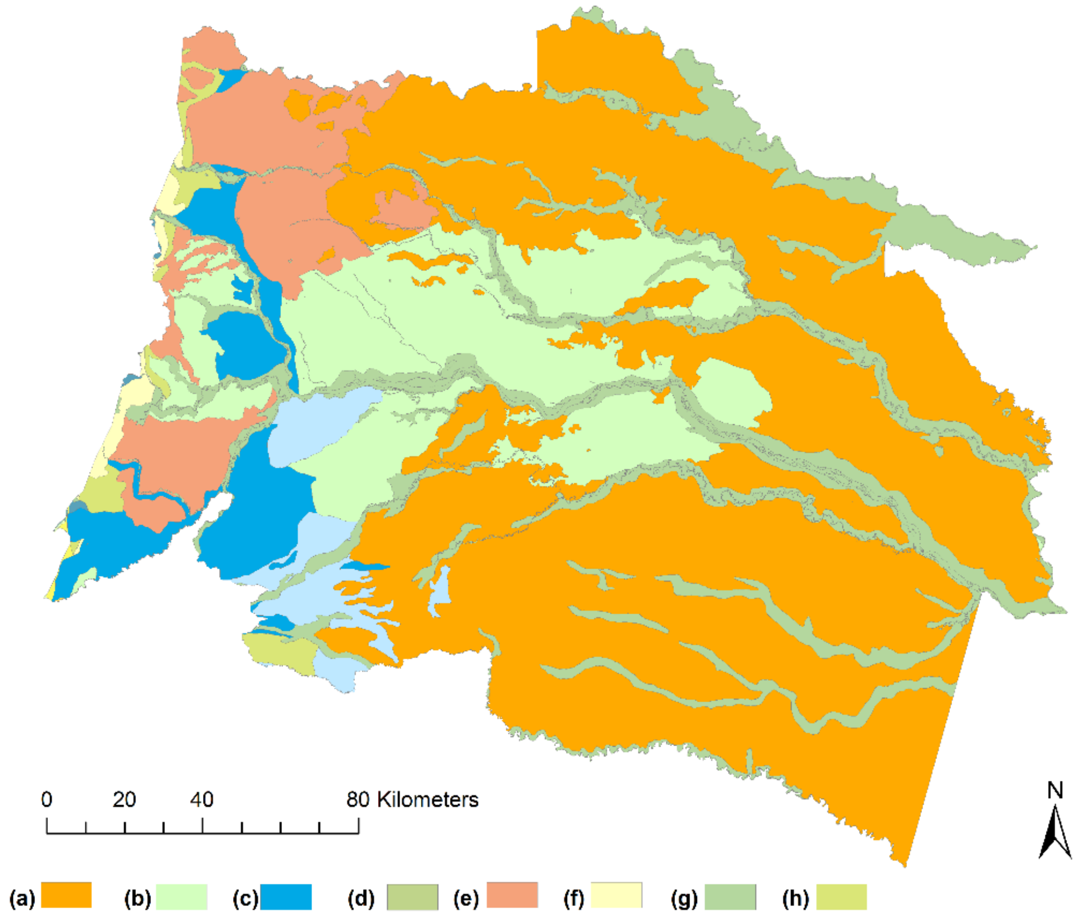


Fig C4. Geological formations (Fm) of what corresponds to the proposed Western Amazon Aquifer System (Rosário et al., 2016):

a) Curaray Fm: Miocene are made of sandy to silty tidalites, argilleux, tuffaceous lutite, fine plastered sandstone.
b) Chambira Fm: Pliocene quartz pebble-bearing conglomerates included in a quartz-rich argillaceous matrix. The basal part contains trough cross-bedded and matrix-supported conglomerates. Main flow directions range from SW–NE to E–W.
c) Miocene Arajuno composed of fine to coarse grained sandstones, to silty tidalites with two drainage directions, WNW–ESE to W–E and N–S.
d) Mesa is composed of clastic deposits from coarser to medium grain size.
e) Mera Fm: Quaternary, thick conglomerates, tuffaceous, sand and argillaceous covered by volcanoclastic deposits, poorly constrained, 45m thickness.

f) Tiyuyacu Fm: Eocene, coarser to fine conglomerate sandy lutites, to mudstones and are organized in typical channel-filling, fining-upward, 10 m-thick sequences. Variable range thickness (150–548 m).

g) Alluvial deposit Fm from the Quaternary.

h) Chalcana Fm from late Oligocene to early Miocene, consists of reddish shales intercalated with rare fine-grained and thin sandstone beds displaying trough cross-bedded stratifications and horizontal laminations with variable thickness (255-455m).

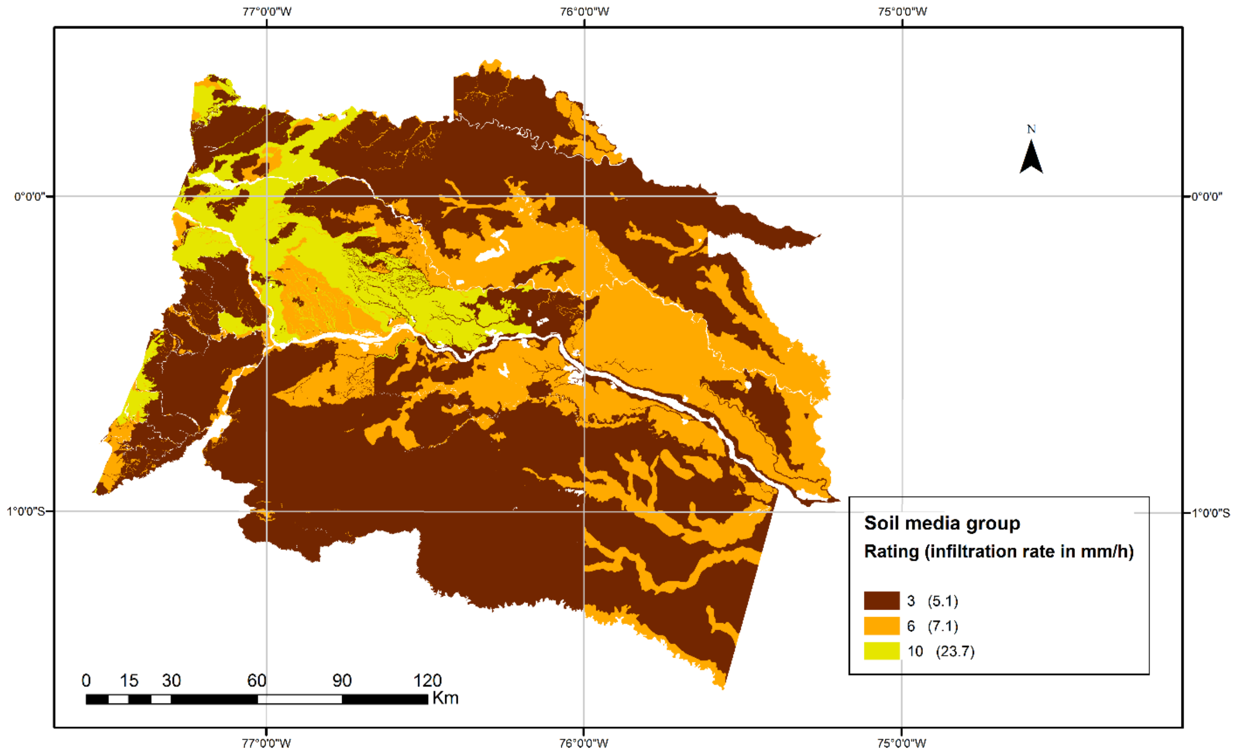


Fig C5. Soil media layers, corresponding to the vadose zone, with soil type and infiltration rate in parentheses.

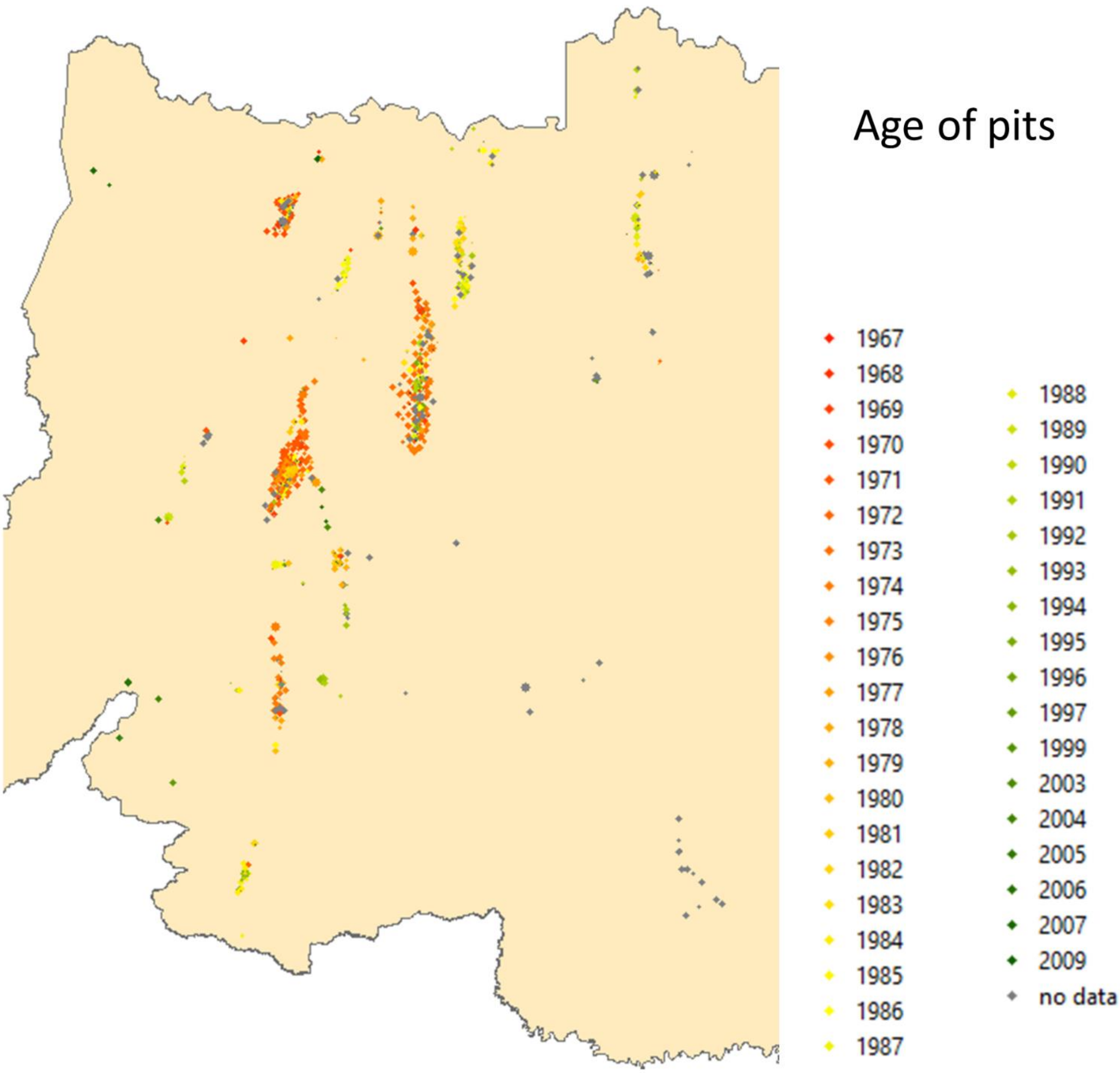


Fig C6. Age of unlined oil pits.

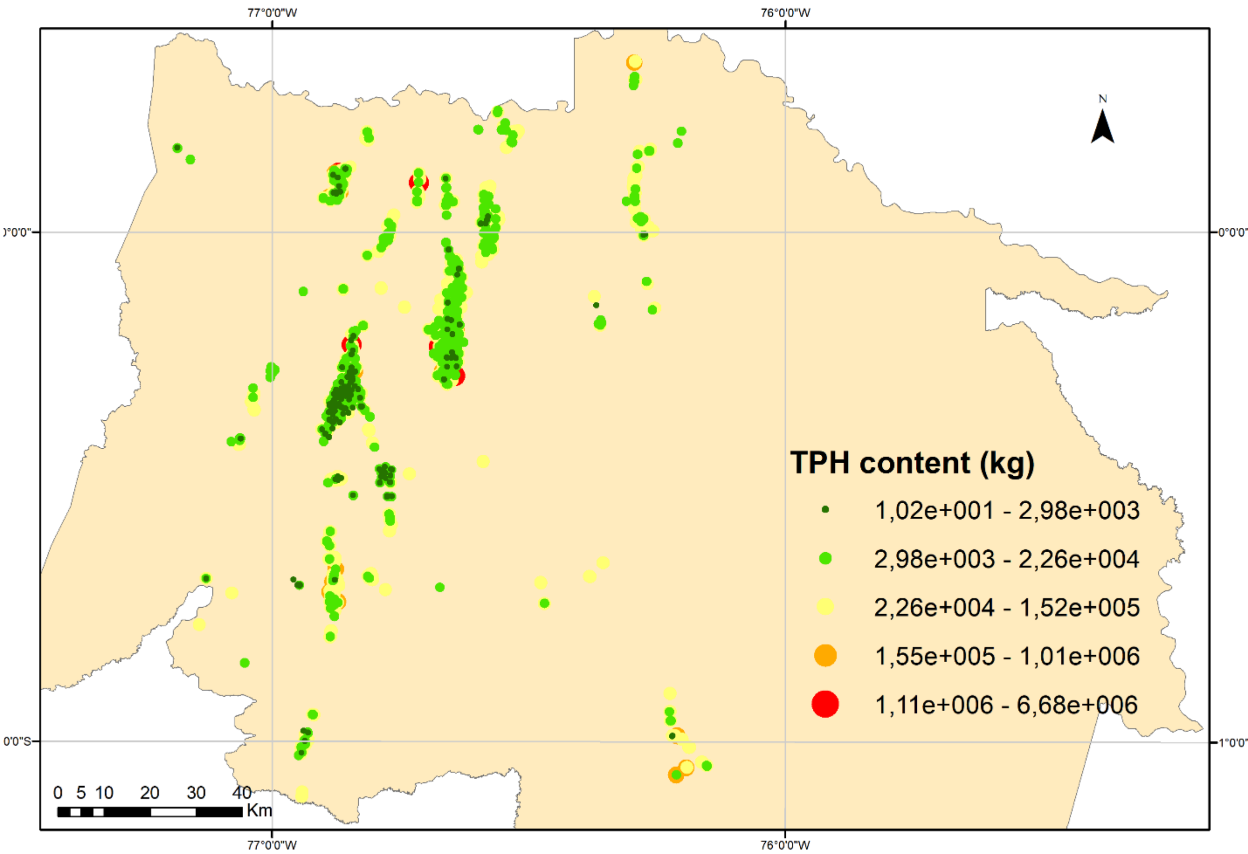


Fig C7. TPH content in unlined oil pits combining original and extrapolated data.

Section D. Density of oil infrastructures by watershed.

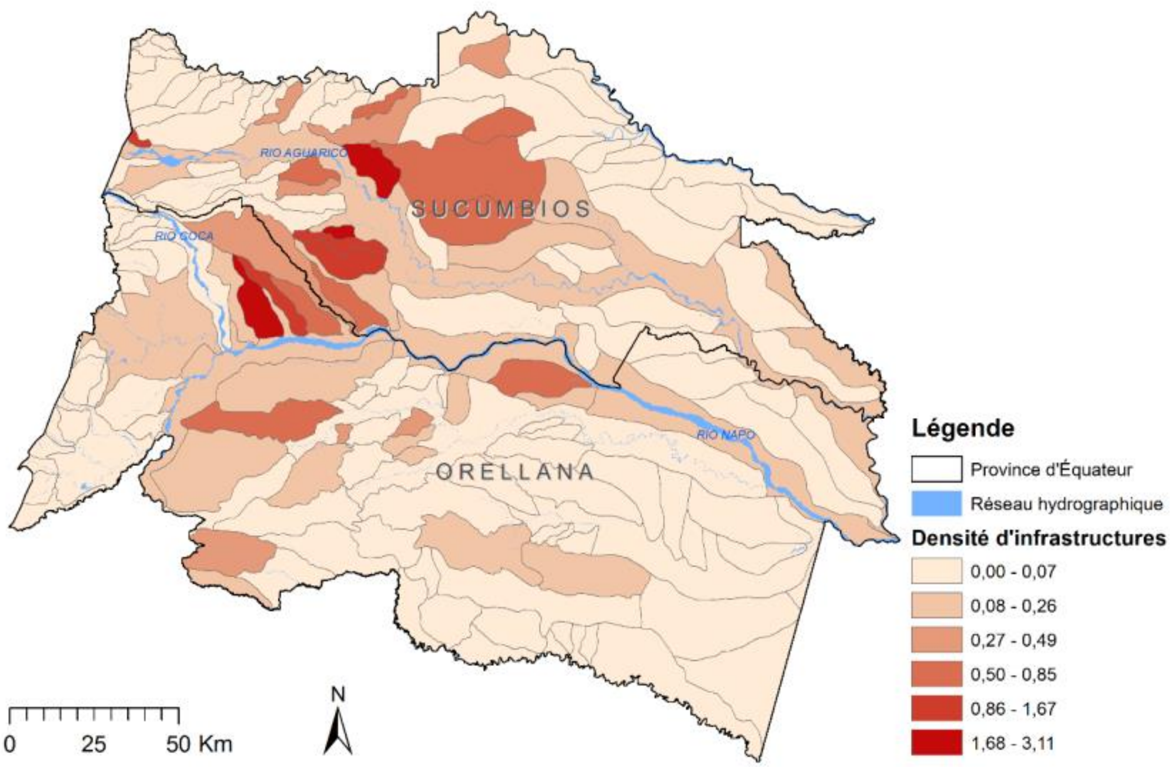


Fig D1. Oil infrastructure density: platforms, stations, flare stacks and mud pits in number per square kilometre, displayed by watershed.